The Regional Implications of Transported Air Pollutants:

An Assessment of Acidic Deposition and Ozone

Volume II—Appendixes

Interim Draft

This interim draft is a product of the ongoing OTA assessment of the "Impacts of Atmospheric Alteration." Available information and analysis is being released at this time to assist congressional deliberations on the Clean Air Act. This draft has neither been reviewed or approved by the Technology Assessment Board. Portions of this material will be contained in the forthcoming OTA Report on transport of air pollutants.

July 1982

CONGRESS OF THE UNITED STATES
Office of Technology Assessment
Washington, D. C. 20510

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The Technology Assessment Board approves the release of this report. The views expressed in this report are not necessarily those of the Board, OTA Advisory Council, or of individual members thereof.

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FOREWORD

Acid rain has been the focus of much debate during the Clean Air Act deliberations of the 97th Congress. The controversy over transported air pollutants -- of which acid rain is one example -- revolves around suspected damage to lakes and streams, forests and soils, agricultural crops, building materials and human health. Some individuals and groups believe that more stringent measures should be taken now by the Congress to control pollutant emissions. Others, pointing to the uncertainties in our knowledge of the causes and consequences of transported air pollutants, are concerned that the costs of such actions may be unwarranted.

Recognizing that the uncertainty surrounding many aspects of the problem complicates the decision of whether, or when, to control transported air pollutants, the Committees responsible for reauthorizing the Clean Air Act asked OTA to evalute the potential implications of transported air pollutants, such as acid rain and ozone. In response to these immediate concerns of Congress, this OTA interim draft focuses on the potential benefits of acting now to abate transported air pollutants versus the potential costs of action that may be premature.

Current understanding of transported pollutants allows a description of potential resource damage -- regions of the U.S. most susceptible to damage -- but not a precise quantification. The costs of likely control strategies, and who might bear these costs, can be described, but the effectiveness of these control strategies for avoiding resource damage cannot be calculated with confidence. Given the difficulty of quantifying the relationship between emissions reductions and the prevention of resource damage, policy decisions may have to be based on the risks of resource damage, the risks of unwarranted control expenditures, and the distribution of these risks among different groups and regions of the country.

Throughout this interim draft, special attention is paid to the regional distribution of risks -- both to those who may bear potentially unnecessary control costs, as well as those who face potential resource damage. This draft is being made available to the 97th Congress to assist its current deliberations on the Clean Air Act. Policy options are not included in this interim draft, but will be presented in the final Report. The Report, to be completed later this year, will also include an updated version of information contained in this draft, additional research results, and a synthesis of findings.

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INTERIM DRAFT

VOLUME 2: APPENDICES

BB Allocation of Sulfur Dioxide Emissions Reductions and the Costs of Control

CC Control Technologies for Reducing Sulfur and Nitrogen Oxide Emissions

DD The Effects of Acid Rain Control on the U.S. Coal Market

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JJ
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3) Indirect Health Effects of Acid Precipitation

MM
1) Interstate Provisions of the Present Clean Air Act
2) Mechanisms for Control of Transboundary Pollution

These appendices are excerpts from completed research papers designed and commissioned by OTA. They present background information, study methodologies, and results in greater detail than the Summaries contained in Volume 1.

Appendix BB

ALLOCATION OF SULFUR DIOXIDE EMISSIONS REDUCTIONS AND THE COST OF CONTROL

In the 31-state region comprised of all states east of the Mississippi, and the first tier of states west of the Mississippi, sulfur dioxide emissions for 1980 are estimated to be about 22.5 million tons. Approximately 70 percent of this total, or 16 million tons, originates from fossil fuel combustion by electric utilities.* Under current regulations, EPA-approved State Implementation Plans require utilities to reduce these emissions by approximately 1.5 million tons, to 14.5 million tons per year. To aid in evaluating proposals to reduce the long-range transport of air pollutants, OTA has developed estimates of the cost to utilities in the 31-state region of reducing sulfur dioxide emissions below the 14.5 million ton SIP-compliance level.

A. OVERVIEW OF MODEL USED IN COST ANALYSIS

OTA's cost estimates have been developed through the use of a computer model that develops cost estimates of reducing emissions at each major utility boiler in the 31-state region (about 2000 boilers in about 900 powerplants).** For each unit, the model calculates the combination of emissions reduction measures that minimizes the costs of complying with seven alternative emissions standards, ranging from SIP compliance to a 0.6 lb./million Btu SO_2 limit. Emissions reductions opportunities can then be ranked on the basis of costs per ton of SO_2 removed. For each alternative standard, the model considers the costs of the following control options for each unit:

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* The slight inconsistencies between the emissions data presented here and in Section A of the report are due to the use of different SO_2 emissions inventories. Estimates of utility emissions regionwide should be considered accurate to within 5-10%.

** The model used for this analysis is the AIRTEST model, run by E.H. Pechan and Associates, Inc. AIRTEST was modified from a larger model used in several earlier major assessments, including the New Source Performance Standards (NSPS) review, the Ohio River Basin Energy Study (ORBES), and the Acid Rain Mitigation Study (ARMS).

For coal-fired powerplant emissions:

- o blending presently-used coals with low-sulfur coals,
- o switching to low-sulfur coals,
- o physical coal cleaning to reduce sulfur contents of presently used coals,
- o installing dry scrubbers in conjunction with either current or alternative coal types, and
- o installing wet scrubbers in conjunction with either current or alternative coal types.

For plants burning residual oil:

- o switching to a lower-sulfur oil.

The model's scrubber cost assumptions are presented in Table BB-1.

The results of this analysis are in turn used to generate state and region-level cost estimates of various levels of sulfur dioxide emissions reductions. These costs can be calculated in three ways:

(1) Regional Least Cost: Emissions reductions opportunities are selected in ascending order of per-ton costs within the entire region until the emission reduction target is achieved. This approach assumes a "perfect market" for the exchange of emissions reductions obligations throughout the entire region, and provides a lower bound model estimate.

(2) State Least Cost: Reduction opportunities are selected in ascending order of per-ton costs within each state until the reduction target is achieved. These estimates assume trading of emissions reductions within states, but not among states.

(3) Plant Cap: Each plant is required to comply with a specified

Table BB-1

RETROFITTED LIMESTONE FGD SYSTEM
TOTAL INVESTMENT COST
6 LB. SO_2/MMBtu - No Spares
(1979 $)

MW	% REMOVAL	INVESTMENT $X 10^6	$/KW
500	90	87.47	175
	70	69.91	140
	50	48.40	97
200	90	42.33	212
	70	32.46	162
	50	28.86	129
100	90	26.75	268
	70	23.13	231
	50	17.80	178

Source: Adapted from MITRE Corporation (1981), "Acid Rain Mitigation (ARM) Study Final Utility and Industrial Boiler FGD Costs and Derived Cost Algorithms".

emissions limit. This approach to estimating costs chooses the least-cost approach at each plant, but assumes no trading of reduction obligations among plants or states.

1. Constraints

The AIRTEST model considers only reductions of sulfur dioxide emissions from utilities. No estimates of the cost of reductions in emissions of nitrogen oxides, nor estimates of the reductions of SO_2 or NO_x from the industrial sector, are calculated. Furthermore, existing utilities are the only sources considered, and are assumed to be operating at their current level of capacity utilization. The OTA analysis assumes that all reductions are accomplished immediately, without accounting for new plants being built, or old plants being retired. Finally, the model considers only a finite group of control alternatives. Potential control options not considered by the model are:

- -- early retirement of major sources;
- -- energy conservation;
- -- selective use of lower-emitting plants; and
- -- advanced control technologies.

C. COST COMPARISON OF ALTERNATIVE APPROACHES TO EMISSIONS REDUCTIONS

Table BB-2 provides data on energy consumption and sulfur dioxide emissions from utilities in the 31-eastern states. The first column presents 1980 emissions levels for each state. The third column presents emission rates, defined in this case as the quantity of sulfur dioxide emitted per million Btu of energy input. In general, the higher the emission rate, the greater the opportunity for emissions reductions and the lower the cost per

TABLE BB-2

EMISSIONS, FOSSIL FUEL CONSUMPTION, AND EMISSION RATES FOR UTILITIES IN THE EASTERN U.S., 1980

	SO_2 Emissions (1000 tons)	Rank	Fossil Fuel Consumed (10^{12} Btu)	Rank	Emission Rate (lb SO_2/mm Btus)	Rank
Alabama	563.7	12	472.9	15	2.3	12
Arkansas	26.5	27	111.1	26	0.5	30
Connecticut	32.2	26	133.2	25	0.5	30
Delaware	52.5	25	71.2	27	1.5	19
District of Columbia	4.6	31	9.5	30	1.0	27
Florida	725.8	10	836.6	3	1.7	18
Georgia	726.3	9	513.6	11	2.9	7
Illinois	1,127.5	5	820.5	4	2.7	9
Indiana	1,536.5	2	733.9	5	4.2	2
Iowa	232.4	17	210.6	[illegible]	2.2	13
Kentucky	983.1	8	561.2	10	3.6	5
Louisiana	25.0	28	489.7	14	0.1	21
Maine	16.3	29	23.0	29	1.4	22
Maryland	223.2	18	206.2	23	2.1	14
Massachusetts	275.5	16	313.6	16	1.8	16
Michigan	565.4	11	614.2	8	1.8	16
Minnesota	180.2	20	234.8	20	1.5	19
Mississippi	132.2	22	199.2	24	1.3	25
Missouri	1,139.7	4	512.0	12	4.5	1
New Hampshire	80.3	24	56.2	28	2.9	7
New Jersey	110.2	23	244.1	18	0.9	29
New York	488.5	13	685.8	7	1.4	22
North Carolina	435.8	15	592.2	9	1.5	19
Ohio	2,168.7	1	1,128.1	2	3.8	3
Pennsylvania	1,441.8	3	1,149.9	1	2.5	11
Rhode Island	5.2	30	10.5	31	1.0	27
South Carolina	219.6	19	219.0	21	1.9	15
Tennessee	933.7	7	506.5	13	3.7	4
Vermont	0.4	32	0.6	32	1.2	26
Virginia	163.2	21	238.1	19	1.4	22
West Virginia	947.1	6	695.8	6	2.7	9
Wisconsin	448.3	14	286.1	17	3.4	6
ARMS Region Total	16,011.4		12,852.9		2.5	

Source: E. H. Pechan & Associates, 1982.

ton of reducing emissions. Table BB-3 displays the average cost of control (in dollars per ton of SO_2 removed) for each state at various levels of reduction. Estimates are shown to vary considerably.

OTA has analyzed a number of alternative approaches to allocating an 8 million ton utility SO_2 emissions reduction to states. The regional costs and distributional consequences -- in terms of the amount of emissions reductions allocated to each state and the cost of achieving those allocated reductions -- are presented below for each of 11 alternative allocation formulae. Table BB-4 presents state-by-state emissions reductions requirements for each formula. Table BB-5 displays the costs to each state, in terms of annual costs per kilowatt-hour (kWh) as a percentage of 1979 electricity rates, of achieving the reductions specified by each formula. The description, rationale, costs, and distributional consequences of each allocation approach are provided below.

TABLE BB-3

COST OF SO_2 REDUCTION BY STATE AND LEVEL OF REDUCTION
(dollars per ton)

	10% Reduction Average	30% Reduction Average	50% Reduction Average
Alabama	275	333	459
Arkansas	427	N/A *	N/A
Connecticut	N/A	N/A	N/A
Delaware	642	672	734
Florida	198	273	393
Georgia	345	372	470
Illinois	108	155	175
Indiana	202	246	270
Iowa	16	55	367
Kentucky	183	243	323
Louisiana	N/A	N/A	N/A
Maine	567	588	N/A
Maryland	151	420	591
Massachusetts	749	1,158	N/A
Michigan	221	233	281
Minnesota	320	483	670
Mississippi	351	351	388
Missouri	40	78	94
New Hampshire	421	577	616
New Jersey	381	494	887
New York	427	490	571
North Carolina	771	891	1,037
Ohio	66	146	209
Pennsylvania	269	377	444
Rhode Island	N/A	N/A	N/A
South Carolina	514	557	648
Tennessee	113	194	275
Vermont	N/A	N/A	N/A
Virginia	326	642	873
West Virginia	114	209	277
Wisconsin	62	114	161

* N/A means not achievable under the assumptions used in the analysis. The model assumes that no plant is required to reduce emissions below a rate of 0.6 pounds SO_2 per MMBtu.

Source: AIRTEST Analyses performed by E. H. Pechan & Associates.

TABLE BB-4

EMISSIONS REDUCTIONS REQUIRED BY VARIOUS ALLOCATION APPROACHES
(reductions in percent below 1980 utility emissions)

	1980 Emissions	SIP Emissions	50% Each	S.1709	1.5 lb Cap	1.3 lb Average	Peak Year	Max. Reduc. Below 1980	Max. Reduc. Below SIP	Fossil Elec.	Total Elec.
Alabama	563.7	454.3	50.0%	38.4%	38.1%	45.5%	43.1%	50.0%	50.0%	44.4%	23.0%
Arkansas	26.5	22.7	50.0	14.6	14.6	14.6	14.6	14.6	14.6	14.6	14.6
Connecticut	32.2	32.2	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Delaware	52.5	52.5	50.0	0.0	20.3	11.8	25.9	30.0	30.0	11.3	29.4
District of Columbia	4.6	4.6	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.2
Florida	725.8	707.9	50.0	37.8	34.0	25.1	37.5	43.5	43.5	24.3	26.7
Georgia	726.3	710.0	50.0	39.3	47.4	54.0	49.9	50.0	51.1	52.3	50.7
Illinois	1,127.5	1,099.5	50.0	55.4	58.4	52.7	54.6	50.0	51.3	54.1	48.4
Indiana	1,536.5	1,373.9	50.0	76.4	65.6	69.0	62.0	50.0	55.3	69.3	72.9
Iowa	232.4	212.1	50.0	8.7	46.6	41.1	44.4	50.0	51.4	45.7	47.2
Kentucky	983.1	744.1	50.0	80.0	57.3	62.9	56.7	50.0	62.2	62.6	66.4
Louisiana	25.0	25.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Maine	16.3	16.3	50.0	0.0	8.8	8.6	13.3	15.4	15.4	11.8	0.0
Maryland	223.2	223.0	50.0	0.1	34.6	39.9	39.4	45.6	45.6	38.1	20.2
Massachusetts	275.5	240.7	50.0	12.6	24.2	26.0	30.9	35.8	35.8	20.3	27.7
Michigan	565.4	517.2	50.0	18.4	31.5	29.4	33.7	39.1	39.1	28.8	25.6
Minnesota	180.2	180.0	50.0	0.1	25.6	15.3	29.8	34.5	34.5	19.2	3.6
Mississippi	132.2	132.2	50.0	42.4	43.8	2.1	44.3	50.0	50.0	2.0	22.4
Missouri	1,139.7	1,139.7	50.0	65.8	68.1	70.8	64.0	50.0	50.0	71.6	74.5
New Hampshire	80.3	67.7	50.0	71.1	47.5	54.4	50.0	50.0	57.9	56.6	57.7
New Jersey	110.2	110.2	50.0	0.0	24.4	0.0	27.7	32.1	32.1	0.0	0.0
New York	488.5	455.2	50.0	15.8	33.0	8.7	37.1	43.0	43.0	9.7	6.8
North Carolina	435.8	435.8	50.0	0.0	4.2	11.7	16.3	19.0	19.0	2.4	8.8
Ohio	2,168.7	1,886.21	50.0	68.7	62.1	66.2	59.7	50.0	56.5	66.3	70.3
Pennsylvania	1,441.8	1,316.0	50.0	53.7	46.4	48.2	48.6	50.0	54.4	47.6	51.3
Rhode Island	5.2	5.2	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
South Carolina	219.6	219.6	50.0	0.0	29.0	35.2	35.3	41.0	41.0	32.1	0.0
Tennessee	933.7	678.2	50.0	81.3	59.3	64.7	58.2	50.0	63.7	63.0	62.5
Vermont	0.4	0.4	50.0	0.0	4.7	0.0	15.0	17.4	17.4	22.6	0.0
Virginia	163.2	163.2	50.0	0.0	3.9	5.1	12.8	14.8	14.8	5.8	0.0
West Virginia	947.1	926.3	50.0	58.8	49.6	52.2	49.2	50.0	51.1	48.9	57.5
Wisconsin	448.3	446.8	50.0	11.3	56.8	58.5	55.0	50.0	50.2	60.5	52.4
ARMS Region Total	16,011.4	14,598.8	50.0%	50.6%	49.8%	50.5%	50.0%	46.8%	50.4%	50.0%	50.2%

TABLE BB-4 (cont.)

EMISSIONS REDUCTIONS REQUIRED BY ALLOCATION APPROACHES BASED ON TOTAL EMISSIONS

(reductions in percent below 1980 total emissions)

	Total 1980 Emissions (1000 tons)	Per Capita	Per Area
Alabama	739.8	14.8%	43.7%
Arkansas	78.9	4.9	4.9
Connecticut	149.1	21.6	0.0
Delaware	90.8	57.8	31.6
District of Columbia	44.4	10.3	0.0
Florida	933.7	1.9	1.9
Georgia	837.7	1.9	32.1
Illinois	1,532.8	34.1	23.5
Indiana	2,079.8	70.2	67.8
Iowa	344.3	5.9	14.1
Kentucky	1,108.0	34.8	61.7
Louisiana	260.5	0.0	0.0
Maine	98.4	0.0	0.0
Maryland	330.0	43.0	0.1
Massachusetts	477.7	57.7	7.3
Michigan	931.7	5.2	5.2
Minnesota	294.0	0.1	0.1
Mississippi	196.1	0.0	0.0
Missouri	1,348.6	6.4	58.6
New Hampshire	109.0	11.6	14.3
New Jersey	362.0	30.4	0.0
New York	1,073.9	17.1	3.1
North Carolina	618.5	0.0	4.6
Ohio	2,802.6	75.3	56.7
Pennsylvania	2,165.6	63.7	41.7
Rhode Island	32.1	16.2	0.0
South Carolina	315.6	0.0	1.1
Tennessee	1,086.0	30.2	53.2
Vermont	15.0	0.0	0.0
Virginia	326.7	0.0	0.0
West Virginia	1,093.6	61.5	75.7
Wisconsin	618.8	0.2	22.1
TOTALS	22,500.7	35.5%	36.1%

Source: Office of Technology Assessment

TABLE BB-[illegible]

STATE-BY-STATE COSTS OF ALTERNATIVE ALLOCATION FORMULAE
(costs as a percentage of 1979 electricity rates)

Percent of 1979 Rates

0% = -
1-3% = *
3-5% = **
5-8% = ***
8-12% = ****
12-16% = *****
16-20% = ******

State	50% EACH	MOYNI-HAN	1.5 LB CAP	1.3 LB AVE	ELEC, TOTAL	PER CAPITA	PER AREA	PEAK YEAR	MAX BASE	MAX SIP
Alabama	*	*	*	*	-	**	-	*	*	*
Arkansas	*	-	-	-	-	-	-	-	-	-
Connecticut	-	-	-	-	-	-	-	-	-	-
Delaware	***	-	*	*	*	***	*****	*	*	*
D.C.	-	-	-	-	-	-	-	-	-	-
Florida	**	*	*	*	*	-	-	*	*	*
Georgia	***	***	***	***	***	**	-	***	***	***
Illinois	*	*	*	*	*	*	*	*	*	*
Indiana	***	*****	****	****	****	******	******	****	***	***
Iowa	*	-	*	*	*	*	-	*	*	*
Kentucky	**	******	***	***	***	****	*	***	**	***
Louisiana	**	-	-	-	-	-	-	-	-	-
Maine	*	-	*	*	-	-	-	*	*	*
Maryland	**	-	*	**	*	-	***	**	**	**
Massachusetts	***	-	*	*	*	-	*****	*	**	**
Michigan	*	*	*	*	*	-	-	*	*	*
Minnesota	***	-	*	*	-	-	-	*	*	*
Mississippi	**	**	**	-	*	-	-	**	**	**
Missouri	**	***	***	***	****	***	*	***	**	**
New Hampshire	***	****	**	***	***	*	-	***	***	**
New Jersey	*	-	*	-	-	-	***	*	*	*
New York	*	*	*	-	-	-	*	*	*	*
North Carolina	****	-	*	*	*	*	-	*	**	**
Ohio	**	***	***	***	***	***	*****	***	**	**
Pennsylvania	**	***	**	**	**	***	****	**	**	***
Rhode Island	-	-	-	-	-	**	-	-	-	-
South Carolina	**	-	*	**	-	-	-	**	**	**
Tennessee	*	*****	**	***	***	***	*	**	*	***
Vermont	-	-	-	-	-	-	-	-	-	-
Virginia	***	-	*	*	-	-	-	*	*	*
West Virginia	***	***	***	***	***	*****	****	***	***	***
Wisconsin	*	*	*	**	*	*	-	*	*	*

1. EQUAL PERCENTAGE REDUCTION IN EACH STATE

Description: Each state is required to reduce its emissions an equal percentage below some baseline level.

Rationale: Each state should share an equivalent burden in the reduction of aggregate emissions. Requiring an equal percentage reduction for each state will equalize the relative emissions reduction.

Formula for Achieving an 8 Million Ton Reduction: 50% reduction below 1980 utility emissions in each state in the 31-state ARMS region.

Cost: Reducing sulfur dioxide emissions by 50% in each state is estimated to cost \$2.6 to \$3.2 billion annually, at an average cost of \$390 to \$480 per ton of SO_2 removed.

Distributional Consequences: This approach to allocation requires an equal percentage reduction from each state regardless of (1) the relative costs of emissions control, or (2) the stringency of the state's existing emissions regulations. Thus, states where relatively less expensive emission reductions are unavailable (high marginal costs of control) -- such as Georgia, North Carolina, and Virginia -- will have the highest relative costs imposed on them.

2. THE TARGETTED RETROFIT

Description: A targetted retrofit selects specific types of existing facilities as the basis for determining emissions reductions. The selection criteria typically are based on the amount of emissions produced by the facility.

Rationale: A relatively small number of sources are responsible for a relatively large amount of emissions. In 1980, the 50 largest emitting plants in the eastern U.S. contributed about 50% of utility emissions. In addition, achieving reductions from the largest emitting facilities could reduce the administrative burden of implementing emissions reductions because reductions would have to be applied at a relatively small number of facilities.

Formula for Achieving an 8 Million Ton Reduction: The Moynihan Bill (S.1709) requires that plants emitting greater than 50,000 tons of sulfur dioxide per year, at a rate greater than 3 pounds of sulfur dioxide per million Btu, must reduce their emissions by 85%. This formula applies to approximately 60 utility plants, and achieves an 8.2 million ton SO_2 reduction from current emissions.

Cost: This allocation approach is estimated to cost $2.5 to 2.9 billion per year, at an average cost of between $370 and $440 per ton of SO_2 removed.

Distributional Consequences: States with large facilities having high emission rates -- such as Ohio, Indiana, Kentucky, and Tennessee -- are those that would incur the greatest emissions reductions and costs. States without "targetted" plants would not have to reduce emissions at all.

3. MAXIMUM EMISSIONS PER UNIT OF ENERGY INPUT (EMISSIONS "CAP")

Description: This approach allocates reductions to states based on some maximum emissions limit (an emissions "cap") for each fossil-fuel electric generating plant. In this case, the limit is an emission rate that would specify the amount of allowable emissions per unit of energy input.

Rationale: Setting an emission cap would allocate the target emission reductions to states with powerplants emitting over a certain rate. It would thus penalize states with plants emitting large quantities of SO_2 per unit of energy input.

Formula for Achieving an 8 Million Ton Reduction: If no plant in the eastern U.S. were allowed to emit at a rate greater than 1.5 pounds of sulfur dioxide per million Btus of energy input, emissions could be reduced by 8 million tons from current levels.

Cost: This allocation approach entails annual costs of $2.0 to $2.7 billion, at an average annual cost of $280 to $370 per ton.

Distributional Consequences: The largest costs and percetage reductions are distributed to the states whose plants emit relatively large amounts of pollutants per unit of energy consumed -- such as Missouri, Indiana, Ohio, and Tennessee (see Table BB-2). States with plants with low emission rates (usually because of the use of less-polluting fuels such as oil and natural gas) -- such as Louisiana, Arkansas, and Connecticut -- are allocated the smallest reductions.

4. AVERAGE EMISSIONS PER UNIT OF ENERGY INPUT

Description: Rather than placing an emissions cap on each facility, reductions could be allocated by requiring that each state achieve a specified average emission rate. Under the average approach, some plants within a state are allowed to exceed the specified emission rate (unlike the cap case) as long as there are compensating plants emitting below the rate.

Rationale: Unlike the cap, the average emission rate approach gives credit to states that have plants below the specified emission rate.

Formula for Achieving an 8 Million Ton Reduction: An 8 million ton sulfur dioxide reduction could be achieved by requiring that each state comply with a state-wide average of 1.3 pounds of sulfur dioxide per million Btus of energy input.

Cost: This strategy is estimated to cost $2.1 to $2.6 billion per year at an average cost of $320 to $400 per ton.

Distributional Consequences: Emissions reductions are allocated in a manner similar to the cap case. States that have a substantial number of plants emitting at rates below the specified average will tend to favor the average over the cap (such as New York, Iowa, and Minnesota); states in which most plants emit at rates well above the average used for allocation (such as Missouri and Kentucky) will tend to favor the cap over the average (assuming identical regional reductions are required).

5. EMISSION CAP -- CREDIT FOR REDUCTIONS WITHIN THE PAST 5 YEARS

Description: This approach is similar to the emissions cap approach, but calculates state-by-state emissions reductions on the basis of the year between 1976 and 1980 in which each state's emissions were the greatest, rather than from 1980 levels.

Rationale: This approach gives states credit for having reduced their emissions in recent years. Because emissions reductions are based on peak year emissions, states that have reduced emissions within the past five years have already achieved a portion of their required reduction.

Formula for Achieving an 8 Million Ton Reduction: A 1.2 lb. SO_2 per MMBtu cap on each state's peak year emissions between 1976 and 1980 can achieve a reduction of about 8 million tons.

Cost: This approach is estimated to cost between $2.1 and $2.8 billion annually, at an average cost of $320 to $430 per ton of SO_2 reduced.

Distributional Consequences: Because this approach is based on an emissions cap, the distribution of reductions and costs is similar to the cap case. However, the requirements of states that have achieved significant emissions reductions within the past five years -- such as Kentucky, Michigan, Minnesota, Ohio, and Tennessee -- will be less.

6. EMISSION CAP -- MAXIMUM REDUCTION OF 50% BELOW CURRENT LEVELS

Description: This approach is similar to a cap but limits the amount which any state can reduce to 50% below 1980 emissions.

Rationale: By placing a ceiling on reductions, this approach lessens the burden placed on those who would have to reduce the most. It is based on the premise that there is a maximum amount beyond which no state should be required to reduce.

Formula for Achieving an 8 Million Ton Reduction: A 1.2 lb. SO_2 per MMBtu cap, with a 50% maximum reduction below 1980 emissions for each state, achieves a 7.5 million ton reduction.

Cost: This approach is estimated to cost between $2.0 and 2.7 billion per year, at an average cost of $330-$440/ton.

Distributional Consequences: By placing a limit on percentage reductions, this approach lessens the burden on those who would be required to reduce the most under a cap approach. The states most favorably affected by this approach are Indiana, Kentucky, Missouri, and Ohio.

7. EMISSION CAP -- MAXIMUM REDUCTION OF 50% BELOW SIP LEVELS

Description: Similar to the previous approach, except that the 50% limit on reductions is calculated from SIP-compliance emissions rather than actual 1980 values.

Rationale: This approach is similar to the 50% maximum reduction applied in #6 above; however, emissions that are out of compliance are not counted when calculating the maximum reduction. Therefore, some states that are not in compliance must reduce emissions by greater than 50%.

Formula for Achieving an 8 Million Ton Reduction: A 1.2 lb. cap, with a maximum reduction of 50% below SIP-compliance levels, achieves an emissions reduction of 8.1 million tons.

Cost: This approach is estimated to cost from $2.3 to $2.9 billion per year, at an average cost of $340-$440/ton.

Distributional Consequences: The allocation of reductions and costs is different from the previous approach only as a result of different degrees of compliance with SIPs. This distinction applies to only those states to which the maximum reduction applies. Of these states, those penalized the most are those most out of compliance -- such as Kentucky, Tennessee, and Ohio.

8. AVERAGE EMISSION RATES -- FOSSIL ELECTRICITY OUTPUT

Description: Emissions reductions are allocated on the basis of the amount of emissions per unit of electricity output from fossil fuel electrical generating plants, rather than energy input in the case of conventional emission rate caps and averages.

Rationale: Gives credit to those states who are efficient in converting fossil fuel energy input into electricity output.

Formula for Achieving an 8 Million Ton Reduction: Reductions are allocated according to an average emission rate of 14 pounds of SO_2 per megawatt-hour of fossil electricity output.

Cost: The costs of this approach range from $2.1 to $2.6 billion per year, at an average cost of $310-$390 per ton of SO_2 reduced.

Distributional Consequences: The allocation of reductions and costs is almost identical to the emission rate average approach (#4), the only difference being variations in the conversion efficiencies of boilers. West Virginia, North Carolina, and South Carolina are shown to be favored by this provision.

9. AVERAGE EMISSION RATES TOTAL ELECTRICITY OUTPUT

Description: This approach would allocate emissions reductions on the basis of the amount of sulfur dioxide emissions per unit of electricity output from all sources, including hydroelectric and nuclear power plants.

Rationale: Allocating emissions reductions on the basis of total electricity output would give credit to those who generate electricity with fuels that do not produce sulfur dioxide emissions.

Formula for Achieving an 8 Million Ton Reduction: Reductions are allocated according to an average rate of 11 pounds of SO_2 per megawatt-hour of total electricity output.

Cost: The costs of this approach are estimated to range from $2.1-2.6 billion annually, at an average cost of $310-$390 per ton of SO_2 reduced.

Distributional Consequences: This approach favors states with relatively high proportions of their electricity produced by hydroelectricity or nuclear power -- such as Alabama, Maine, New York, Wisconsin, Maryland, and Virginia.

10. AVERAGE EMISSION RATE: LAND AREA

Description: This approach allocates emissions on the basis of emissions densities, i.e. the amount of sulfur dioxide emitted per unit of land area. Since this formula does not rely on inputs to or outputs from the electricity generating process, all emissions can be considered, rather than just utility emissions.

Rationale: To the extent that acid deposition is produced by local sources, limiting the concentration of SO_2 emissions might help limit the concentration of acidic deposition in the surrounding area.

Formula for Achieving an 8 Million Ton Reduction: Reductions are allocated according to an average rate of 140 pounds of SO_2 per hectare.

Cost: This approach would cost $3.4 to $3.5 billion per year, at an average cost of $520 to $540 per ton of SO_2 removed.

Distributional Consequences: The states with the highest emissions densities, and thus the largest reductions and costs under this approach, are D.C., Ohio, Massachusetts, Indiana, and Pennsylvania.

11. AVERAGE EMISSION RATE: POPULATION

Description: Emissions reductions are calculated on the basis of the amount of pollution emitted per person residing in the state. Reductions are based on total emission from the region, not just utility emissions.

Rationale: This approach to allocating emissions gives credit to those states that -- because of their reliance on clean fossil fuel combustion, non-SO_2-emitting electricity generation, higher energy efficiency, or fewer SO_2-producing industrial activities -- have lower emissions to population ratios.

Formula for Achieving an 8 Million Ton Reduction: Reductions are allocated according to an average rate of 200 pounds of SO_2 per capita.

Cost: The costs of this approach are estimated to range from $2.6 to $2.9 billion annually, with an average cost of $390 to $540 per ton of SO_2 reduced.

Distributional Consequences: Those states with a relatively high proportion of total sulfur dioxide emissions to the population supported by the economic activities that generate those emissions are allocated the largest reductions. These states include West Virginia, Indiana, Kentucky, Missouri.

D. THE IMPACTS OF A NATIONAL ALLOCATION APPROACH

OTA has analyzed the effect of expanding the 31-state ARMS region to encompass the entire 48 contiguous states. Table BB-6 compares the emissions reduction requirements in the 31-state and 48-state regions for one type of allocation approach: an emissions cap of 1.5 pounds of SO_2 per million Btus. Utility SO_2 emissions in 1980 in the 31-eastern states amounted to 92% of the national utility total (excluding Alaska and Hawaii). Because emissions rate are in general lower for the western states, expanding the region under consideration results in only marginal increases in total emissions reductions. Under the national approach, the 31-eastern states account for 99% of the required emissions reductions.

F. COMPARISON OF MODEL AND UTILITY ESTIMATES OF THE COSTS OF S. 1706 (THE MITCHELL BILL

Table BB-7 compares available analysis of S. 1706 - a 10 million ton reduction is sulfur dioxide emissions - performed by utilities with OTA estimates of the cost impact of this bill on specific states. Because of differences in assumptions of the costs of scrubbing, low-sulfur coal, and money for capital investment, the utility estimates are consistently higher. This information has been compiled for the Senate Committee on Environment and Public Works, and is in the process of being evaluated by OTA and other Congressional agencies. A more detailed analysis of the differences between utility and OTA and other model estimates will be presented in the final OTA report.

TABLE BB-6

COMPARISON OF NATIONAL AND REGIONAL APPROACHES TO EMISSION CONTROLS

	1980 Utility Emissions (1000 tons)	SIP Emissions (1000 tons)	Required Reductions (1000 Tons)	Percent Below Current
Alabama	563.7	454.3	214.6	38.1
Arizona	88.7	87.4	0.0	1.4
Arkansas	26.5	22.7	0.0	14.6
California	78.6	59.3	0.0	0.4
Colorado	80.7	80.7	0.3	0.0
Connecticut	32.2	32.2	0.0	24.6
Delaware	52.5	52.5	10.7	20.3
District of Columbia	4.6	4.6	0.0	0.0
Florida	725.8	707.9	247.0	34.0
Georgia	726.3	710.0	344.0	47.4
Idaho	0.0	0.0	0.0	0.0
Illinois	1,127.8	1,099.5	658.2	58.4
Indiana	1,536.5	1,373.9	1007.2	65.6
Iowa	232.4	212.1	108.3	46.6
Kansas	144.1	119.8	53.2	36.9
Kentucky	983.1	744.1	563.1	57.3
Louisiana	25.0	25.0	0.0	0.0
Maine	16.3	16.3	1.4	8.8
Maryland	223.2	223.0	77.3	34.6
Massachusetts	275.5	240.7	66.6	24.2
Michigan	565.4	517.2	178.1	31.5
Minnesota	180.2	180.0	46.1	25.6
Mississippi	132.2	132.2	57.9	43.8
Missouri	1,139.7	1,139.7	775.6	68.1
Montana	28.3	28.3	2.7	9.5
Nebraska	49.4	49.4	2.5	5.1
Nevada	39.3	39.3	0.0	0.0
New Hampshire	80.3	67.7	38.2	47.5
New Jersey	110.2	110.2	26.9	24.4
New Mexico	99.5	99.1	0.2	0.4
New York	488.5	455.2	161.2	33.0
North Carolina	435.8	435.8	18.2	4.2
North Dakota	118.0	118.0	12.7	10.7
Ohio	2,168.7	1,886.2	1347.0	62.1
Oklahoma	37.6	37.6	0.0	0.0
Oregon	3.3	3.2	0.0	0.0
Pennsylvania	1,441.8	1,316.0	668.4	46.4
Rhode Island	5.2	5.2	0.0	0.0
South Carolina	219.6	219.6	63.8	29.0
South Dakota	28.5	28.5	3.8	13.4
Tennessee	933.7	678.2	554.0	59.3
Texas	329.0	329.0	0.0	0.0
Utah	38.8	38.8	0.0	0.1
Vermont	0.4	0.4	0.0	4.7
Virginia	163.2	163.2	6.4	3.9
Washington	69.3	69.3	8.2	11.8
West Virginia	947.1	926.3	470.1	49.6
Wisconsin	448.3	446.8	254.7	56.8
Wyoming	119.7	79.5	0.6	33.6
NATIONAL TOTAL	17,364.4	15,773.4	8049.3	46.3
ARMS REGION TOTAL	16,011.4	14,598.8	7695.1	49.8

TABLE BB-7

COMPARISON OF MODEL AND UTILITY ESTIMATES OF S. 1706

State and Utility	Percent Rate Increases Utility Estimate	 OTA State Estimate**
Illinois		5 - 8
Central Illinois Public Service	19%	
Indiana		12 - 14
Public Service of Indiana	50	
Hoosier Energy	62	
Indianapolis Power and Light	49	
Indiana & Michigan Electric Company*	56 - 89	
Kentucky		9 - 12
Kentucky Power Co.*	55 - 87	
Missouri		13 -19
Union Electric	50	
New Hampshire		7 - 8
Public Service of New Hampshire	15	
Ohio		7 - 8
Cincinnati Gas and Electric	18	
Columbus and Southern Ohio Electric Co.*	38 - 77	
Ohio Power Company*	65 - 87	
Ohio Edison	40	
Tennessee		8 - 10
Tennessee Valley Authority	8 - 13	
Kingsport Power Company*	55 - 59	
West Virginia		7 - 10
Wheeling Electric Company*	57 - 76	
Appalachian Power Company*	45 - 81	
Wisconsin		7 - 8
Wisconsin Public Service Commission	4 - 17	
Wisconsin Eletric Power Company	20	
Wisconsin Power and Light	12	

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* Members of the American Electric Power system. Lower estimate is for residential rates, upper estimates for industrial rates. The average impact over the entire system is 63%.

** OTA estimates are based on the AIRTEST model, run by E. H. Pechan and Associates. Percent rate increases reflect estimated costs of S. 1706 in 1979 dollars over 1979 electricity rates.

Appendix CC

CONTROL TECHNOLOGIES FOR REDUCING SULFUR AND NITROGEN OXIDE EMISSIONS

PART I GENERAL OVERVIEW

Acid deposition and ozone are believed to result primarily from the chemical transformation of three pollutants: oxides of sulfur, oxides of nitrogen, and hydrocarbons. This report discusses the techniques available for reducing oxides of sulfur and nitrogen. Where possible, for each emission control approach, the report will describe:

- the processes involved in the technique;
- its stage of development, i.e., whether the technology is currently commercially available or requires further research and development;
- the effectiveness of the technique, i.e., the degree of reduction it can reliably achieve;
- costs; and
- secondary effects.

The major source of NO_x and SO_x is the combustion of fossil fuels. During the combustion process, sulfur contained in the fuel reacts with oxygen to form sulfur dioxide gas (SO_2) and, after reaction with other elements, sulfate particles. Nitrogen contained in both the air used for combustion as well as in the fuel reacts with oxygen to form gaseous nitrogen oxides and, after reaction with other elements, nitrate particles. There are three general approaches to controlling these emissions:

- Pre-combustion: the amount of sulfur or nitrogen in the fuel being burned can be reduced, either by using fuels naturally lower in sulfur or nitrogen content, or by subjecting the fuels to some kind of physical or chemical process to remove sulfur and nitrogen;
- During combustion: the combustion process can be altered to reduce

the amount of sulfur and nitrogen compounds released in the gas stream;

- Post-combustion: the products of combustion can be treated to remove pollutants before they are released into the atmosphere.

All three of these approaches have been successfully utilized to reduce emissions from existing sources.

In addition to these differences among approaches to control, the stage of development of the emissions control techniques discussed in this report varies considerably. Technologies may be characterized as:

- *In-use* technologies -- those being sold on a commercial scale in the U.S. that have demonstrated control capabilities.
- *Available* technologies -- those that have been tested and proven but are not currently operational in the U.S. on any significant scale, even though they may be widely utilized in other countries.
- *Emerging* technologies -- those still in the research and development phase, but that have been tested on at least a pilot scale.[1]

Control approaches also differ widely in the amount of emissions reductions they are capable of achieving and in their cost-effectiveness. Each technology is most cost-effective in a particular range of emissions reductions. For instance, the pre-combustion approach of physical coal cleaning is only technically feasible for sulfur dioxide reductions of less than 40 percent. On the other hand, the post-combustion approach of flue-gas desulfurization is cost-effective in the 50 to 95 percent sulfur dioxide removal range. The control technique appropriate for a given facility thus depends a great deal on the level of reduction desired.

Table 1 presents summary information on the control technologies described in this report. Due to site-specific conditions, the removal efficiency levels given are intended for approximation only.

TABLE 1

OVERVIEW OF CONTROL TECHNOLOGIES

Control Technology	Reduction Efficiencies	Revenue Requirements (Mills/Kwh)	Stage of Development
Sulfur Dioxide			
Fuel Switching	30-90%	0-7	In Use
Physical Coal Cleaning	5-40%	1-5	In Use
Chemical Coal Cleaning	60-85%		Emerging
Wet Flue Gas Desulfurization	70-95%	10-17	In Use
Dry Flue Gas Desulfurization	40-90%	9-15	In Use
Regenerable Flue Gas Desulfurization	70-90%	12-25	Available
Oil Desulfurization			
Indirect	30-40%		In Use
Direct	70-90%		In Use
Limestone Injection Multi-stage Burner	50-90%		Emerging
Fluidized Bed Combustion	<90%		Emerging
Nitrogen Oxides			
Low-NO_x Burner - Commercial	30-50%	0	In Use
Low-NO_x Burner - Developmental	50-80%	0	Emerging
Flue Gas Treatment Without Catalyst	35-40%		Available
Flue Gas Treatment With Catalyst	80-90%		Available

PART 2 CONTROLLING SULFUR DIOXIDE EMISSIONS

I. PRECOMBUSTION APPROACHES

A. Fuel-Switching

Sulfur dioxide is formed when sulfur, an element naturally present in fossil fuels, is oxidized during the combustion process. The greater the concentration of sulfur in the fuel being burned, the greater the production of sulfur dioxide gas. One way to reduce sulfur dioxide emissions is therefore to use fuels with lower concentrations of sulfur.

The amount of emissions reductions attainable by fuel-switching at a given plant depend upon: (1) the amount of sulfur in the fuel currently being used, and (2) the amount of sulfur in the fuel available for replacement. The sulfur content of coal currently being used for electricity generation varies considerably, from about 0.2 to 5.5 percent sulfur by weight.[2] An analysis of the emissions reduction potential at the 50 largest-emitting plants in the U.S. shows that switching from high- to low-sulfur coal can achieve between 25 to 85% reduction in emissions. If all of these 50 plants switched to low-sulfur coals, annual sulfur dioxide emissions from these facilities could be reduced by about 6 million tons, or about 65 percent.[3] This is equivalent to a 27 percent reduction of total sulfur dioxide emissions in the eastern U.S. Greater emissions reductions could be achieved if, in addition, fuel-switching were applied to other plants.

The two principal components of the costs of fuel-switching are: (1) the "fuel-price differential", i.e., the difference in price between the high- and low-sulfur fuel, and (2) the type of fuel-handling facilities, boilers, and emissions control devices at the plant. Low-sulfur coal is typically more expensive than high-sulfur coal, especially in the east and midwest.

Figures 1 and 2 illustrate the cost differences between high- and low-sulfur coal. Figure 1 shows the costs of a high-sulfur, Illinois coal at various distances away from the mine. Similar costs are presented for an Appalachian low-sulfur coal in Figure 2. The costs of the low-sulfur coal are shown to be 50% higher than the high-sulfur coal even in the areas closest to the mines. Cost differentials are maintained as the destination of the coal moves farther from the mine. The cost differential confronting specific utility plants will vary considerably, depending on site-specific factors and market arrangements.

Many western low-sulfur coals contain more ash than eastern high-sulfur coals. Additionally, flue gas from boilers fired with low-sulfur fuel requires larger precipitators than gas from those fired with high-sulfur fuel, due to changes in the ash resistivity caused by SO_2. Therefore, particulate emissions control devices (electrostatic precipitators or baghouses) generally have to be upgraded if low-sulfur fuels are used at existing plants. Fuel-handling facilities may also have to be altered because western coals are often more difficult to pulverize than eastern coals. In addition, certain kinds of boilers are designed to burn coal with very specific characteristics (such as energy yield, ash, and moisture content). These boilers would have to be derated or modified to burn low-sulfur coal efficiently. The capital costs of upgrading particulate controls, fuel-handling facilities, and boilers are relatively minor compared to the increased fuel costs involved in fuel-switching. One estimate of the cost of achieving a 6 million ton sulfur dioxide emissions reduction by fuel-switching at the 50 largest emitters is

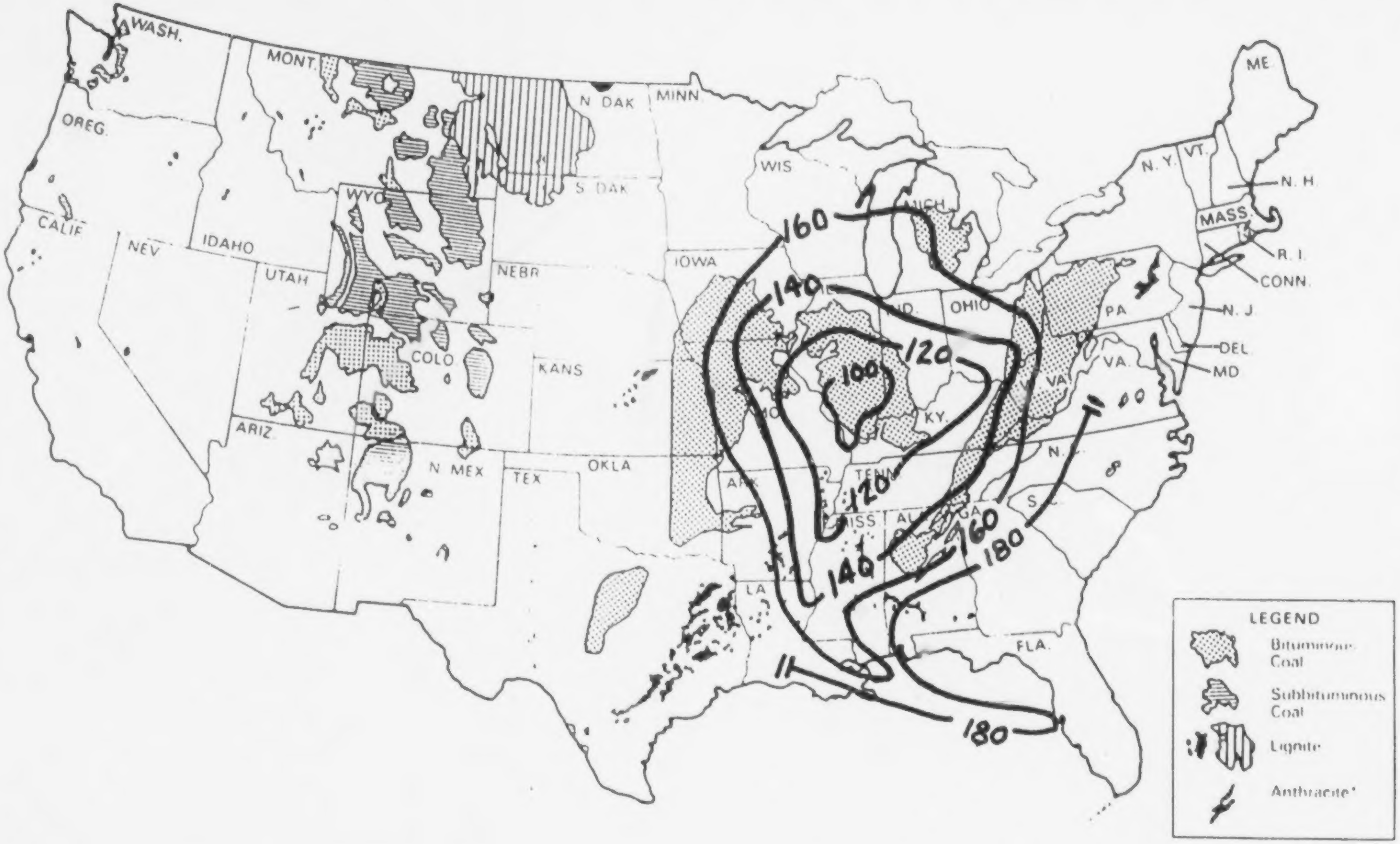
WASH.
OREG.
CALIF
NEV
IDAHO
MONT.
WYO
UTAH
ARIZ.
COLO
N. MEX
N DAK
S. DAK
NEBR
KANS
OKLA
TEX
MINN
IOWA
WIS
MICH
ILL
IND.
OHIO
KY.
TENN
MO
ARK
LA
MISS
ALA
GA
FLA.
S. C.
N. C.
VA.
W. VA.
PA
N. Y.
VT.
N. H.
ME.
MASS.
R. I.
CONN.
N. J.
DEL
MD
100
120
120
140
140
160
160
180
180
LEGEND
Bituminous Coal
Subbituminous Coal
Lignite
Anthracite

Figure 1 : Iso-cost curves for Illinois coal 1980.
Delivered prices in nominal cents per million Btu.

Source: FPC Form 423

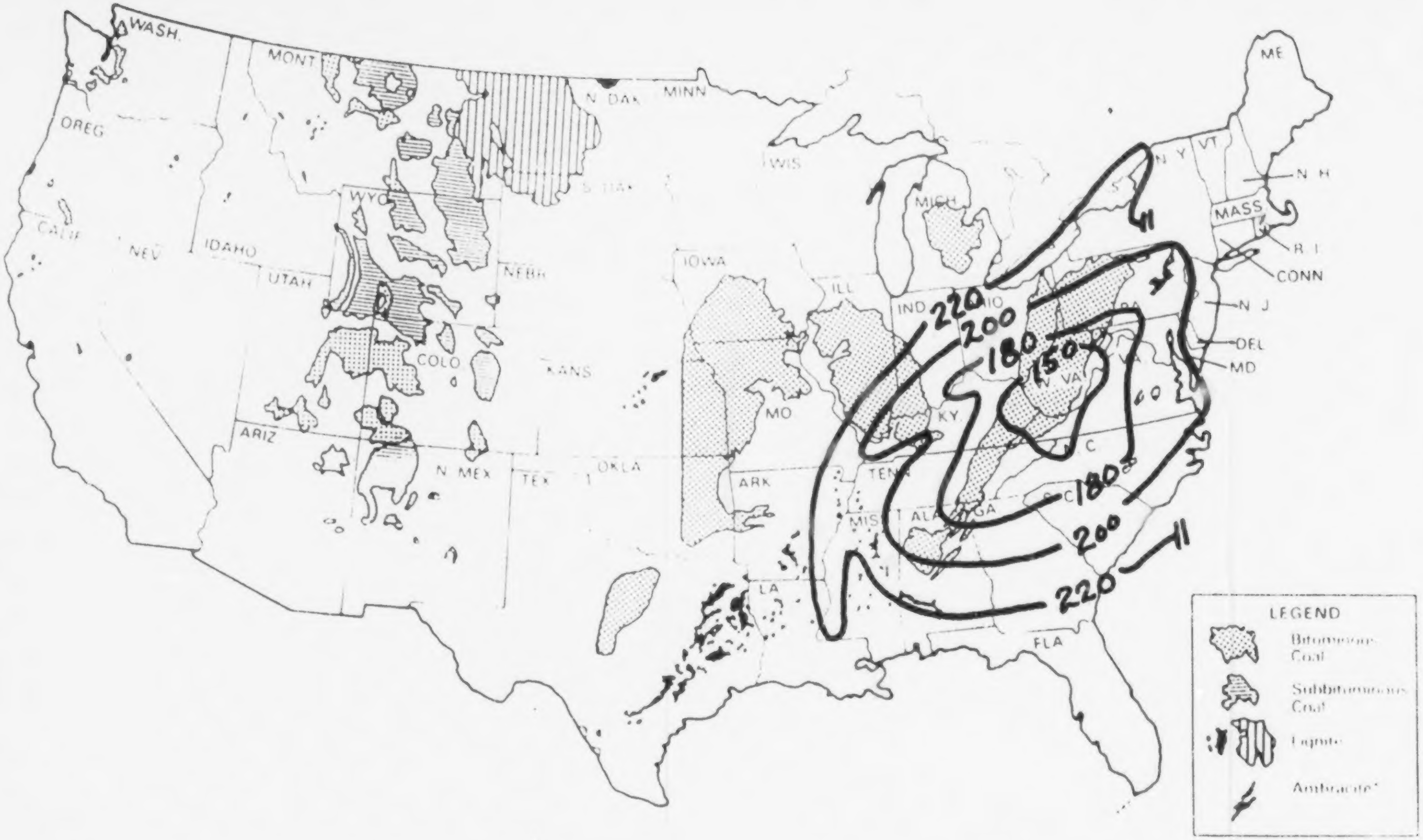
WASH.
OREG
CALIF
NEV
IDAHO
MONT
UTAH
WYO
ARIZ
COLO
N MEX
N DAK
S DAK
NEBR
KANS
OKLA
TEX
MINN
IOWA
WIS
MO
ARK
LA
ILL
MICH
IND
OHIO
KY
TENN
MISS
ALA
GA
FLA
S C
N C
VA
W VA
PA
N Y
VT
N H
ME
MASS
R I
CONN
N J
DEL
MD
220
200
180
150
180
200
220
LEGEND
Bituminous Coal
Subbituminous Coal
Lignite
Anthracite

Figure 2 : Iso-cost curves for E. Kentucky/W. Virginia Low Sulfur* coal, 1980. Delivered prices in nominal cents per million Btu.

*NOTE: Low Sulfur specified as less than 1.2 pounds SO_2 per million Btu.

Source: FPC Form 423

$1.2 billion per year (1980 dollars), or $210 per ton of sulfur dioxide removed.[5]

There are 2.17×10^{11} tons of "compliance" coal (i.e., those capable of meeting a 1.2 lb/MMBtu sulfur dioxide emissions limit without the application of control technologies) in the demonstrated reserve base.[6] This quantity of coal could support U.S. production for a period of 50 years (assuming 50 percent recoverability and 3 percent annual growth in consumption). Therefore, achieving substantial emissions reductions through fuel-switching would not be constrained by the resource base over the near future.

However, there are logistical and social constraints to large-scale emissions reductions through fuel-switching. The logistical problems refer primarily to the capacity of the production system and transportation network. Eighty-five percent of the nation's "compliance" reserves are located west of the Mississippi, while 63 percent of coal consumption and 72 percent of coal production occurs east of the Mississippi.[7] In order to be viable, a large-scale emissions reduction program relying primarily on fuel-switching would have to significantly expand western coal production and transportation capacity. The social constraints arise from the community stress caused by rapid expansions in economic activities in western communities and rapid contractions in eastern high-sulfur coal-mining communities (see Appendix DD).[8]

While logistical and social difficulties may make sole reliance on fuel-switching to achieve large-scale emissions reductions an unattractive strategy, fuel-switching in combination with other emissions control approaches can be a significant and cost-effective means of control.

B. Coal Cleaning: Physical or Chemical

The second precombustion approach involves physically or chemically treating coal to remove some of the sulfur it contains. Sulfur in coal exists in two major forms: inorganic and organic. Inorganic or "pyritic" sulfur can be removed relatively inexpensively by exploiting differences in the physical properties of pyrite and coal particles. Organic sulfur is chemically bound to the carbon molecules of coal, and can only be removed by breaking the bonds through some chemical process. These chemical processes are less developed and more expensive than the physical processes that remove pyritic sulfur, but their sulfur removal potential is higher.[9]

1. Physical Coal Cleaning

Physical coal cleaning (often called coal washing) takes advantage of differences in the sizes, densities, and surface properties of pyrite and coal particles. The first step of the cleaning process is to separate raw coal into different size ranges. Breakers and crushers are used to separate the softer coal from the harder rock and other debris contained in the coal entering the treatment plant. After breaking and crushing, the coal is typically filtered through screens to divide it into coarse, intermediate, and fine size ranges.

The method used to extract the pyritic sulfur from the coal depends on the size of particles. Coarse and intermediate size particles allow differences in the specific gravity of pyritic sulfur and coal particles to be used (pyrite has a specific gravity of 5 -- i.e. is 5 times heavier than water; coal is approximately 1.4). The coal mixture is immersed in a fluid, where the heavier pyritic particles sink and the lighter particles float. The coal product and refuse material can then be removed separately.

Fine mineral particles cannot be effectively separated by the specific gravity techniques. Physical coal cleaning of fine particles relies on a process in which the raw coal particles are treated with a chemical that, because of differences in surface properties, adsorbs differently on the surface of coal particles than other substances in the mixture. Air bubbles introduced into the chamber attach to the coal particles and carry the coal to the surface, where they can be skimmed off. The pyrite and other particles sink, and are removed separately.

Table 2 shows the extent to which coal produced for the utility market by eight eastern and midwestern coal-producing states is physically cleaned. One-third of the coal produced by these states was washed in 1979, resulting in an estimated 1.8 million ton reduction in the potential sulfur dioxide emissions (approximately equivalent to a 10 percent reduction in the potential sulfur dioxide emissions from the coal produced by these eight states).[10] Thus, physical coal cleaning is currently a widely used technique.

The emissions reduction potential of physical coal cleaning depends primarily on: (1) the initial sulfur level in the raw coal, (2) the ratio of pyritic to organic sulfur, and (3) the coal-cleaning technique used. Pyritic sulfur accounts for between 30 and 70 percent of the total sulfur content of coal.[11] Higher-sulfur coals tend to have a larger proportion of pyritic sulfur than lower-sulfur coals. Consequently, the higher the sulfur content of coal, the greater the percentage removal possible through this process. Table 3 shows the results of an analysis prepared for EPA on the potential for sulfur removal by coal cleaning in eight eastern and midwestern states.[12] As the first column of the table shows, reductions of between 8 and 33 percent

TABLE 2

CURRENT (1979) SO_2 REDUCTION ACHIEVED BY CLEANING UTILITY COAL FROM EIGHT STATES

Region and State in Which Coal Was Mined	Coal Delivered to Utilities in 1979 (10^3 tons)	Utility Coal Cleaned in 1979 (percent)	Sulfur Content of Coal (expressed as 10^3 ton SO_2) As Mined (10^3 tons)	Sulfur Content of Coal (expressed as 10^3 ton SO_2) As Delivered (10^3 tons)	Average SO_2 Reduction by Coal Cleaning in 1979 (percent)
Northern Appalachia					
Pennsylvania	47,400	30	2,100	1,860	12
Ohio	38,300	11	2,750	2,670	3
Northern West Virginia	31,300	23	1,760	1,690	4
Southern Appalachia					
Southern West Virginia	17,500	9	300	290	1
Virginia	13,400	7	280	270	1
Eastern Kentucky	68,600	22	1,630	1,570	4
Eastern Midwest					
Western Kentucky	38,100	34	2,880	2,600	13
Indiana	25,300	52	1,620	1,410	13
Illinois	49,500	72	3,570	2,780	22
Alabama					
Alabama	14,600	32	460	440	5
EIGHT-STATE TOTAL	344,000	33	17,350	15,580	10

Source: Versar, Inc., Coal Resources and Sulfur Emission Regulations: A Survey of Eight Eastern and Midwestern States, prepared for EPA, PB 81-240319, May 1981.

TABLE 3

AVERAGE EMISSION POTENTIALS AND EMISSION REDUCTIONS FOR COAL FROM EIGHT STATES

Region and State	Average Emission Reduction Using Physically-Cleaned Coal* (%)	Average Emission Potential (lb $SO_2/10^6$ Btu)	Number of Washability Samples
Northern Appalachia			
Pennsylvania	33.2	4.0	170
Ohio	25.9	5.8	90
Northern West Virginia	28.9	4.9	30
Southern Appalachia			
Southern West Virginia	10.1	1.4	16
Virginia	7.6	1.1	8
Eastern Kentucky	15.9	2.3	13
Eastern Midwest			
Western Kentucky	31.5	6.6	37
Indiana	26.4	5.9	21
Illinois	29.3	6.6	40
Alabama			
Alabama	10.8	2.0	10

* Coal crushed to 1/12 inch top size and separated at 1.6 specific gravity.

Source: Versar, Inc., Coal Resources and Sulfur Regulations: A Survey of Eight Eastern and Midwestern States, prepared for EPA, PB81-240319, May, 1981.

are attainable. The second column lists the sulfur dioxide emissions rate (in lbs./million Btu) achievable after coal washing.

Table 4 shows the costs and emissions reduction potential of requiring all coal to be cleaned before its use. An additional reduction in sulfur dioxide emissions of about 2.9 million tons could be achieved from the coals produced by these eight states (equivalent to an 18 percent reduction of emissions from coal produced by these states). As the table shows, coal cleaning is associated with a wide range of costs -- from a low of $310 per ton of SO_2 removed in western Kentucky to a high of $4,100 per ton removed in southern West Virginia (in 1978 dollars). Cleaning high-sulfur coals -- those with the largest emissions reduction potential -- is in general more cost-effective than cleaning lower-sulfur coals. The regionwide average cost is shown to be $710/per ton of SO_2 removed. Coal cleaning adds between $5 and $10 per ton to the price of coal, and between 2.2 mills/kwh and 4.4 mills/kwh in annual revenue requirements.[13] This compares with an average price of residential electricity of 47 mills/Kwh.[14]

DOE's contractor has assessed the costs and emissions reduction potential of washing the coal delivered to the 50 largest emittors in the U.S. Sulfur dioxide removal efficiencies range from 3 to 34 percent. Costs range from $4 to $7 per ton of coal cleaned, $170 to $4900 per ton of sulfur dioxide removed, and from 0.8 to 4.4 mills/kwh (1980 dollars). Cleaning the coal used by these 50 plants is estimated to yield a 1.5 million ton reduction in sulfur dioxide emissions (about 7 percent of total SO_2 emissions in the eastern U.S.) at an average annual cost of $870 million, or $580 per ton of SO_2 removed.[15] This study does not account for the emissions reductions or costs of coal used by these utilities that is currently being cleaned.

TABLE 4

ADDITIONAL SO_2 REDUCTION THAT COULD HAVE BEEN ACHIEVED IF ALL THE 1979 COAL FROM EIGHT STATES HAD BEEN CLEANED

Region and State in Which Coal Was Mined	Additional SO_2 Reduction[1] by Washing All Coal		Levelized Cost of Cleaning the Additional Coals		Cost Effectiveness of Coal Cleaning
	(10^3 tons)	(percent)	(\$/ton[2])	(mills/kWh[3])	(\$/ton SO_2 Removed)
Northern Appalachia					
Pennsylvania	470	25	9.10	4.0	670
Ohio	740	28	9.30	4.0	430
Northern West Virginia	280	16	8.00	3.5	720
Southern Appalachia					
Southern West Virginia	30	11	7.90	3.4	4,100
Virginia	30	10	8.30	3.5	3,700
Eastern Kentucky	260	16	10.00	4.4	2,200
Eastern Midwest					
Western Kentucky	530	21	6.50	3.1	310
Indiana	180	13	4.60	2.2	310
Illinois	230	8	6.60	3.3	430
Alabama					
Alabama	70	17	8.60	3.8	1,200
EIGHT-STATE TOTAL	2,820	18	8.40	3.8	710

1. Over current practice.
2. Of raw coal.
3. For a generating unit with a heat rate of 10,000 Btu/kWh.

Source: Versar, Inc., Coal Resources and Sulfur Emission Regulations: A Survey of Eight Eastern and Midwestern States, prepared for EPA, PB 81-240319, May 1981.

Coal cleaning has several benefits in addition to reduced sulfur dioxide emissions. First, cleaning reduces the ash content of coal, reducing ash disposal requirements at the power facility. Second, the removal of impurities (sulfur, ash, and others) increases the "heating value" (energy per unit of weight) of coal. Increased heating value reduces coal transportation costs and pulverization requirements at the plant. Finally, because cleaning creates a fuel with more uniform characteristics (e.g., ash, moisture, sulfur, and energy content), increased efficiency of boiler operation is possible.[16] The cost benefits can in many instances offset the costs of coal cleaning.

Physical coal cleaning may produce a substantial amount of solid waste. The cleaning process causes approximately one-fourth of the mined material to be discarded as waste. Moreover, some coal (approximately 5 to 10 percent of the energy value) is lost in the process of removing impurities.[18]

2. Chemical Coal Cleaning

Chemical coal cleaning can remove higher percentages of sulfur contained in coal because it can in some cases remove organic as well as pyritic sulfur. These processes, however, have only been successfully operated at the laboratory scale, and are estimated to be five to 10 years away from commercial viability. Chemical coal-cleaning processes vary widely from relatively simple methods that use chemical solutions to leach sulfur and other impurities out of coal, to processes such as Solvent Refined Coal (SRC), which alters the characteristics of coal so much that it is usually considered a coal-conversion process.[19]

Two of the chemical coal cleaning processes receiving the greatest amounts of current research attention are the Meyers process and microwave desulfurization. The Meyers process, developed by TRW, is a chemical leaching process that combines coal with a ferric sulfate or sulfuric acid solution to remove sulfur. This process can remove 80 to 99 percent of the pyritic sulfur in coal (larger removal efficiencies than physical coal cleaning), but cannot remove organic sulfur.[20]

One of the several coal cleaning processes that remove organic as well as pyritic sulfur is microwave desulfurization. Developed by General Electric, this process begins by wetting crushed coal with a sodium hydroxide solution; the mixture is then briefly irradiated with microwave energy. During irradiation, the sodium hydroxide reacts with pyritic and organic sulfur to form sodium sulfide. The coal is immersed in water to remove the sulfur-laden sodium sulfide, and the process is repeated again. Laboratory tests have achieved total sulfur removals in excess of 90 percent.[21]

Because chemical coal-cleaning techniques are in the early stages of development, costs are difficult to estimate. It is not yet clear whether chemical coal cleaning will be economically competitive with flue-gas desulfurization in the future. Chemical coal cleaning can be expected to produce the same side effects -- waste disposal requirements, removal of ash and other impurities, increased heating value, and improved boiler efficiency -- as physical coal cleaning.

C. Oil Desulfurization

Oil desulfurization is a widely applied method for reducing sulfur dioxide emissions. In oil desulfurization, oil is treated with hydrogen, which partially removes the sulfur by combining with it to form hydrogen sulfide gas. One method of desulfurizing oil is hydrodesulfurization. In this method, oil is distilled to separate the crude into various petroleum products. Most of the sulfur concentrates in the heavier residues. The lighter fractions, or distillate, are redistilled under a vacuum, then hydrotreated (i.e., reacted with hydrogen) to remove the sulfur as hydrogen sulfide gas. The product is finally reblended with the vacuum residue to yield low-sulfur fuel oil. This method can reduce the sulfur content by 30 to 42 percent.[23]

In another variation of hydrodesulfurization, the residue from distillation is hydrotreated, and then reblended with the distillate to form a lower-sulfur fuel, or both the residue and distillate from vacuum distillation are separately hydrotreated before reblending. This technique can achieve a degree of desulfurization as high as 70 to 90 percent. Hydrodesulfurization is presently the most prominent method for producing low-sulfur (less than 1 percent sulfur) fuel oil from high-sulfur crudes.[24] Both techniques are similar to coal cleaning in that the higher the degree of desulfurization, the higher the costs.

Disadvantages of hydrodesulfurization are the high investment and operating costs, and the high energy requirements. In the future, since fuel oil combustion is not expected to increase significantly in the U.S., the present desulfurization capacity is expected to remain at current levels.

II. COMBUSTION ALTERATION APPROACHES

A. Limestone Injection Multistage Burner (LIMB)

Many in the U.S. consider the Limestone Injection Multistage Burner (LIMB) to be one of the most promising control technologies under development today. The technique might significantly reduce both SO_x and NO_x emissions.

LIMB is based on the use of staged burner techniques for NO_x control, in combination with sorbent (normally limestone) which is injected through the burners for SO_x control. Sulfur dioxide reacts with the limestone to form solid calcium sulfate.

This technology is still under development; its removal efficiencies and costs are very uncertain at this time. Planning goals established by EPA set objectives of 50 to 70 percent removal of SO_x and NO_x, at a capital cost of $30-40/kw.[25] If these goals are achieved, LIMB would offer substantial cost improvements over existing technologies. It is very possible that LIMB, because it may be retrofitted into existing plants at a competitive cost, may emerge as a particularly attractive control technology option.[26] However, EPA plans to limit the LIMB research program to basic bench- and large pilot-scale R&D through 1985, since funding is unavailable for government sponsorship of a full-scale demonstration at this time.

B. Fluidized Bed Combustion

Another technique that removes sulfur dioxide during the combustion process is fluidized bed combustion. For this process, crushed coal is fed into a bed of inert ash mixed with limestone or dolomite. The bed is held in suspension ("fluidized") by the injection of air from the bottom of the bed. Sulfur dioxide, formed during combustion, reacts with the limestone or dolomite to form solid calcium sulfate, which can be removed from the boiler without interrupting the combustion process.

Fluidized bed combustion can remove up to 90 percent of the sulfur dioxide. However, the costs are very uncertain at this time. Available estimates show the cost-effectiveness of fluidized bed combustion to be about equal to conventional boilers using flue-gas desulfurization.[27]

Aside from lower emissions, fluidized bed boilers have the advantages of greater energy efficiency and smaller unit size than conventional boilers.

III. POST-COMBUSTION APPROACHES

A. Flue Gas Desulfurization

Flue gas desulfurization (FGD) technology removes the sulfur dioxide produced during combustion by spraying the exhaust gases in the stack with a chemical absorbent, typically lime or limestone. This process is popularly referred to as "scrubbing." Of the three types of FGD systems -- wet, dry, and regenerable -- wet processes are most widely used.

1. Wet Scrubbers

The most common absorbents used for wet scrubbing are lime and limestone. The absorbent is dissolved or suspended in water to form a slurry that can then be sprayed or forced into contact with escaping gases. The slurry converts sulfur dioxide into calcium sulfite and calcium sulfate (gypsum) solids.

Technology to wet-scrub the flue gas with a lime or limestone slurry has been commercially available for about 10 years. Wet scrubbing is currently the most widely used type of FGD system in the U.S. As of March 1981, 5.1% of installed generating capacity (and 14% of coal-fired capacity) was controlled by wet scrubbers. By 1990, the figure is projected to increase to 9.4% of installed capacity.[28]

Wet lime or limestone scrubbers can remove between 70 to 90 percent of the sulfur dioxide formed during combustion. With the addition of another chemical -- adipic acid -- removal efficiencies can be increased to 95 percent, while limestone requirements can be reduced by up to 15 percent.[29] However, adipic acid additives may present additional sludge disposal problems.

A TVA study conducted in 1980 estimated the capital costs of a wet limestone system using low-sulfur western coal (0.7 percent sulfur, 9,700 Btu/lb) to be $168-176/kW. For high-sulfur eastern coal (3.5 percent sulfur, 11,700 Btu/lb) capital costs range from $236-244/kW. These estimates are based on costs for a new 500 MW plant, operating at a 63 percent lifetime capacity. The range in cost estimates is due to variations in bids from different contractors and the specific considerations for each site. Annual revenue requirements from the TVA study range from 10.5 to 10.9 mills/kWh for low-sulfur coal and 16.4 to 16.7 mills/kWh for high-sulfur coal.[30]

The costs presented thus far are for flue-gas desulfurization units integrated into new plants. Retrofitting scrubbers onto older plants is about 30 percent more expensive. The principal design concerns are space limitations and higher rates of excess air, necessitating larger units. The actual space required for scrubber equipment is not large, but it is important that it be located near the existing stack. Otherwise, expensive, lengthy duct systems would be required.[31]

Operating problems associated with wet systems are corrosion/erosion of metal surfaces, scaling (where hard sulfate and sulfite deposits form on equipment), and plugging (where soft deposits form). Ways of minimizing these problems are currently being researched. In addition, operation of wet scrubbers requires approximately 3 to 5 percent of a plant's energy output.[32]

The major environmental disadvantage of wet FGD systems is that they produce large amounts of sludge. Limestone scrubbing produces a compound (mainly calcium sulfite and sulfate) that has the consistency of toothpaste, making it difficult to dewater, store, and handle. The total amount of FGD waste produced in a typical 1,000 MW plant burning 3.5 percent sulfur coal is about 225,000 tons annually. A recent report concluded that in the future the U.S. will produce more sludge from FGD scrubbing than from treating municipal sewage.[33]

Sludge may, however, be chemically treated to reduce its water content and improve its compressive strength. Forced oxidation converts the waste calcium sulfite to calcium sulfate, which precipitates as large crystals with better settling characteristics. Other means of improving the sludge's properties, such as fixation with lime and fly ash, are still being developed.

Another problem associated with sludge disposal is the leaching of toxic metals from the residual fly ash into nearby ecosystems. EPA is currently conducting research on the characteristics of leaching of metal compounds from sludge disposal sites to evaluate the seriousness of the problem.[34]

2. Dry Processes of FGD

Dry scrubbers are one of the newest and fastest-growing segments of the FGD market. The process involves injection of a lime slurry or soda ash solution into a spray dryer concurrently with the flue gas. The lime or sodium carbonate reacts with the SO_2 to form a dry, solid product which is subsequently collected along with the fly ash in an electrostatic precipitator (ESP) or fabric filter (baghouse).

Dry scrubbers offer several advantages over wet scrubbing. Although they generate more waste than wet systems, they produce a dry waste product that is easier to handle and recycle than wet sludge, and involve simpler equipment, less maintenance, lower capital costs, and lower energy requirements. In addition, dry systems require less water than wet systems, and thus are especially desirable in western areas of the U.S. where water supplies are limited.[35]

There are, however, some disadvantages to using a dry scrubber over a wet system. First, dry systems require lime, which is more expensive than limestone. Second, dry scrubbers are in the early stages of commercialization and have not demonstrated as high a degree of removal as wet scrubbers. Their use has generally been limited to medium- and low-sulfur coals. However, pilot demonstration and commercial plant tests have shown sulfur removal efficiencies exceeding 90 percent for high-sulfur coal.

As of August, 1981, only two dry scrubber systems were in operation, both at industrial plants, and one utility scrubber was in the start-up phase. By 1983, 13 additional dry systems are expected to be in operation, 9 of which are for utility plants.[36] A TVA study estimates that capital costs for dry flue-gas desulfurization systems range from $168 to 176/kW for low-sulfur western coal, and from $180 to 188/kW for high-sulfur eastern coals. Annual revenue requirements are estimated to range from 8.7 to 9.1 mills/kWh for western low-sulfur coal, and 14.5 to 14.9 mills/kWh for high-sulfur coal. These annual costs are between 10 and 17 percent lower than wet systems, as reported by the same TVA study. An EPA survey of dry systems sold to utilities, however, suggests that actual capital costs might be lower. Reported capital costs for these systems range from $80 to 130/kW.[37]

3. Regenerable Processes

Major research efforts by various government agencies have gone into regenerable FGD processes, which combine the SO_2 in powerplant flue gases with chemicals to produce a marketable product. Eight regenerable FGD systems are currently operating in the U.S., accounting for about 8 percent of the total FGD-controlled electricity generation. The most prominent regenerable FGD process in use in the U.S. is the Wellman-Lord process. It involves scrubbing the exhaust gas with sodium sulfite solution, resulting in sodium sulfite-bisulfite, which is then heated to give off concentrated SO_2 gas that can be used to produce either sulfuric acid or elemental sulfur. This process is already in use by the New Mexico Public Service Company. One disadvantage of this process is its high energy requirements, which are approximately 8 to 12 percent of boiler energy input.[38]

Other regenerable systems under development are the Magnesia scrubbing process, which produces sulfuric acid, and the Rockwell process, which produces sulfur.

PART 3 CONTROLLING NITROGEN OXIDE EMISSIONS

Oxides of nitrogen are formed during combustion by two processes. Like sulfur dioxide, NO_x are formed as a result of the oxidation of nitrogen present in the fuel ("fuel NO_x"). NO_x are also formed by the oxidation of nitrogen in the surrounding air ("thermal NO_x"). Both processes are controlled by the amount of oxygen present, while the thermal NO_x formation is also controlled by temperature. The proportion of thermal to fuel NO_x produced during combustion varies from fuel to fuel. For coal, EPRI estimates that 20 to 40 percent of NO_x emissions are "thermal" and 60 to 80 percent are "fuel."[39]

NO_x emissions, being dependent on the amount of oxygen present and the temperature of the combustion process, can be most directly controlled by modifying combustion conditions. While the majority of NO_x control techniques focus on the combustion process, post-combustion techniques (flue-gas treatment) are also being developed. One pre-combustion technique, the denitrogenation of fuel oil, is being researched, but will not be discussed because of its early stage of development and limited potential.

I. COMBUSTION MODIFICATIONS

Thermal NO_x formation can be minimized by regulating the combustion temperature through delayed mixing of fuel and air in the combustion chamber. Limiting fuel NO_x is somewhat different, requiring control of the fuel-air ratio throughout the entire combustion process. Two of the major techniques used in combustion modification, low excess air (LEA), and low-NOx burners are presented below. Other combustion modification techniques include: off-stoichiometric combustion through biased firing, overfire air, flue gas recirculation, low air preheat, and water injection.[40]

LEA involves reducing the combustion air to the minimum amount required for total combustion. Thus, less oxygen is available for the formation of both thermal and fuel NOx. LEA requires no new hardware, and coupled with other techniques can achieve further emissions reductions merely through changes in operating practices. Also, the reduced airflow causes the quantity of the gas released per unit to decrease, resulting in an improvement in boiler efficiency.

The second-generation, low-NO_x burners under development, which incorporate features of the staged combustion process, have been shown to significantly reduce the formation of both fuel and thermal NO_x[41] in experimental systems and limited boiler applications. During the first stage of combustion, less air is supplied to the burner than is required to completely burn the fuel. Fuel-bound nitrogen is then released, but is converted into N_2 because it cannot be oxidized. The subsequent addition of air causes the remaining fuel to be burned.

The amount by which NO_x emissions can be reduced depends upon very site-specific factors, including the type of fuel burned, the type of boiler in use and whether the plant is new or already exists. Retrofitted onto an existing coal-fired plant which does not control NO_x emissions, the low-NO_x burner can reduce NO_x emissions by as much as 50 percent. Potential NO_x emissions reductions from retrofitting an oil-fired burner range from 60 to 80 percent.[42]

The low-NO_x burner can achieve emissions reduction at relatively low cost. Capital costs for coal-fired plants are approximately 1-5 $/KW if integrated into new boilers, and 2-10 $/KW if retrofitted onto existing plants.[43]

Potential problems such as corrosion and high maintenance requirements could delay large-scale use of the low-NO_x burner. Reducing the amount of air in the first stage of combustion may lead to accelerated corrosion of boiler walls; however, the surrounding secondary air used with these burners is postulated to minimize this potential problem. Retrofitting old boilers can be difficult, but NO_x controls on new boilers can be made an integral part of boiler design without adding substantially to cost.

Another combustion modification approach for the control of NO_x is the Limestone Injection Multistage Burner (LIMB), which is discussed in further detail in the section on combustion alteration approaches for SO_x. EPA research goals for LIMB are to achieve a 50 to 70 percent removal of SO_x and NO_x, at a cost of $30-40/KW; however, LIMB is not expected to be commercially available before 1985.[44]

II. POST-COMBUSTION APPROACHES

A. Flue-Gas Treatment

Flue-gas treatment (FGT) is an emerging post-combustion process for high levels of NO_x removal. FGT has been developed and applied extensively in Japan for use on oil-fired boilers. But due to its operational complexities and high costs for use on coal-fired boilers, FGT has not become as popular as the low-NO_x burner in the U.S.

At least 50 different types of flue-gas treatment technologies are available today. Several of them, termed selective catalytic reduction (SCR), have proven that higher reductions can be obtained through the use of ammonia than from other FGT technologies. SCR is a dry process, produces no solid waste, and in most cases is retrofittable. In SCR, flue gases are mixed with ammonia (NH_3) and then passed over a catalyst. The catalyst assists in the reaction of ammonia and nitrogen oxides to form nitrogen gas and water vapor. While 90 percent NO_x removal during combustion is possible, 80 percent removal is preferrable in order to minimize capital and operating costs and maximize the burners' reliability and life span. One estimate places the costs of FGT at between $75 and $100/KW for a 60 to 80% reduction in NO_x emissions.[45]

Two problems associated with SCR are the disposal of spent catalysts, such as vanadium and titanium, and the condensation of bisulfate and bisulfite residuals onto downstream equipment.[46]

FOOTNOTES FOR CONTROL TECHNOLOGY

1. This typology is used to characterize technologies in Workgroup 3B, Emissions, Costs, and Engineering Assessment, Phase I, Submitted in Fulfillment of the MCI Signed by the U.S. and Canada.

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4. E.H. Pechan and Associates, material supplied to OTA, October 1981.

5. PEDCo, p. 3-2.

6. E.H. Pechan and Associates, information supplied to OTA, April 1982.

7. DOE/EIA-0118(79), Coal Production -- 1979, April 30, 1981.

8. See Office of Technology Assessment, Direct Use of Coal, April 1979; OTA, An Assessment of Development and Production Potential of Federal Coal Leases, December, 1981, especially Chapter 12. For an account of the socio-economic effects of the decline of coal production, see T.R. Ford, ed., The Southern Appalachian Region, A Survey, (Lexington, Ky.: University of Kentucky Press, 1960).

9. For a survey of coal cleaning techniques, see James D. Kilgroe, "Coal Cleaning for Sulfur Dioxide Emission Control", paper submitted to the Acid Rain Conference, Springfield, Va., April 8-9, 1980. The description of physical coal cleaning in this section relies heavily on Kilgroe's paper. Other surveys of coal cleaining techniques can be found in EPA-450/3-81-004, Control Techniques for Sulfur Oxide Emissions from Stationary Sources, Second Edition, April 1981; and EPA-600/7-78-002, Engineering/Economic Analyses of Coal Preparation with SO_2 Cleanup Processes, January 1978.

10. Versar, Inc., Coal Resources and Sulfur Emission Regulations: A Survey of Eight Eastern and Midwestern States, prepared for EPA, PB81-240319, May 1981.

11. Kilgroe, p. 3.

12. Versar.

13. Ibid.

14. Energy Information Administration/Department of Energy, Annual Report to Congress, Volume II, p. 170.

15. PEDCo, Tables 1 and 3-6.

16. EPA-600/7-78-002, pp. 24-33. See also, EPA-600/7-80-105. Cost Benefits Associated with the Use of Physically Cleaned Coal, prepared by PEDCo, May, 1981.

17. Kilgroe, p. 9.

18. EPA-450/3-81-004, p. 4.2-15. See also PEDCo, May 1981, Table 3-6.

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20. EPA-600/7-79-178C, Technology Assessment Report for Industrial Boiler Applications: Coal Cleaning and Low Sulfur Coal, prepared by Versar, Inc., December, 1979.

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23. Elam, N. and Trichen Consultants, Ltd., Present and Future Levels of Sulfur Dioxide Emissions in Northern Europe, Swedish Ministry, June, 1979, p. 98.

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26. Personal communication with Julian Jones, EPA, Research Triangle Park.

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29. Workgroup 3B.

30. Burnett, T.A. et al., "Spray Dryer FGD: Technical Review and Economic Assessment", presented at U.S. Environmental Protection Agency Sixth FGD Symposium, Houston, Texas, October 28-31, 1980.

31. EPA-450/3-81-004, p. 4.2-2.5.

32. Ibid.

33. U.S. Environmental Protection Agency, Sulfur Emissions: Control Technology and Waste Management, Decision series, Office of Research and Development, May 1972, p. 12.

34. Workgroup 3B.

35. EPA-450/3-81-004, p. 4.2-16.

37. EPA-600/7-81-097.

38. EPA-450/3-81-004.

39. EPRI Journal, January/February, 1982, p. 20.

40. EPA, An Analysis of the Economic Incentives to Control Emissions of Nitrogen Oxides from Stationary Sources, January, 1981, p. A3.

41. EPA-600/7-79-178f.

42. Ibid., p. 1-8.

43. EPA-600/7-79-178f, Table 4-2; EPA, An Analysis of the Economic Incentives to Control Emissions of Nitrogen Oxides from Stationary Sources, p. A-7.

44. Abbott and Martin, see footnote 25.

45. EPRI Journal, January/February, 1982, p. 20.

46. Mobley, J.D., Assessment of NO_x Flue Gas Treatment Technology, U.S.EPA, Research Triangle Park, presented at Symposium of Stationary Combustion NO_x Control, Denver, Colorado, October, 1980.

Appendix DD

THE EFFECTS OF ACID RAIN CONTROL MEASURES ON THE U.S. COAL MARKET

I: OVERVIEW

Many legislative proposals for controlling acid rain have focused on reducing sulfur dioxide emissions in the eastern United States. By significantly increasing the value of low-sulfur coals as compared to high-sulfur coals, emissions reductions designed to control acid rain create the potential for significant shifts in the coal market. Because the sulfur content of coal varies significantly from region to region throughout the United States, any emission reduction strategy that relies even in part on "switching" to lower sulfur coals can potentially affect the regional distribution of coal production and employment. These changes in the coal market could be accompanied by social dislocations as some areas experience rapid economic growth and others, declines.

This analysis is organized in the following manner:

- Part II discusses the characteristics of U.S. coal reserves and production as a basis for the potential patterns of change in regional coal production as a result of potential acid rain control measures.
- Part III assesses the potential magnitude of changes in regional coal production resulting from acid rain control measures, and the economic consequences of these changes. OTA uses as the basis for its estimates results of a computer model that projects region-by-region changes in coal production due to sulfur dioxide emissions reductions of 5, 10, and 13 million tons. OTA has combined these production projections with coal-miner productivity and income data to project the effects of acid rain controls on regional coal employment and economic activity. These estimates will then be compared to similar analyses performed by Peabody Coal and the United Mine Workers of America.

- Part IV addresses the availability and efficacy of societal measures to mitigate potential social disruption caused by regional shifts in coal production due to acid rain controls.

II: FACTORS CONTRIBUTING TO POTENTIAL COAL MARKET SHIFTS

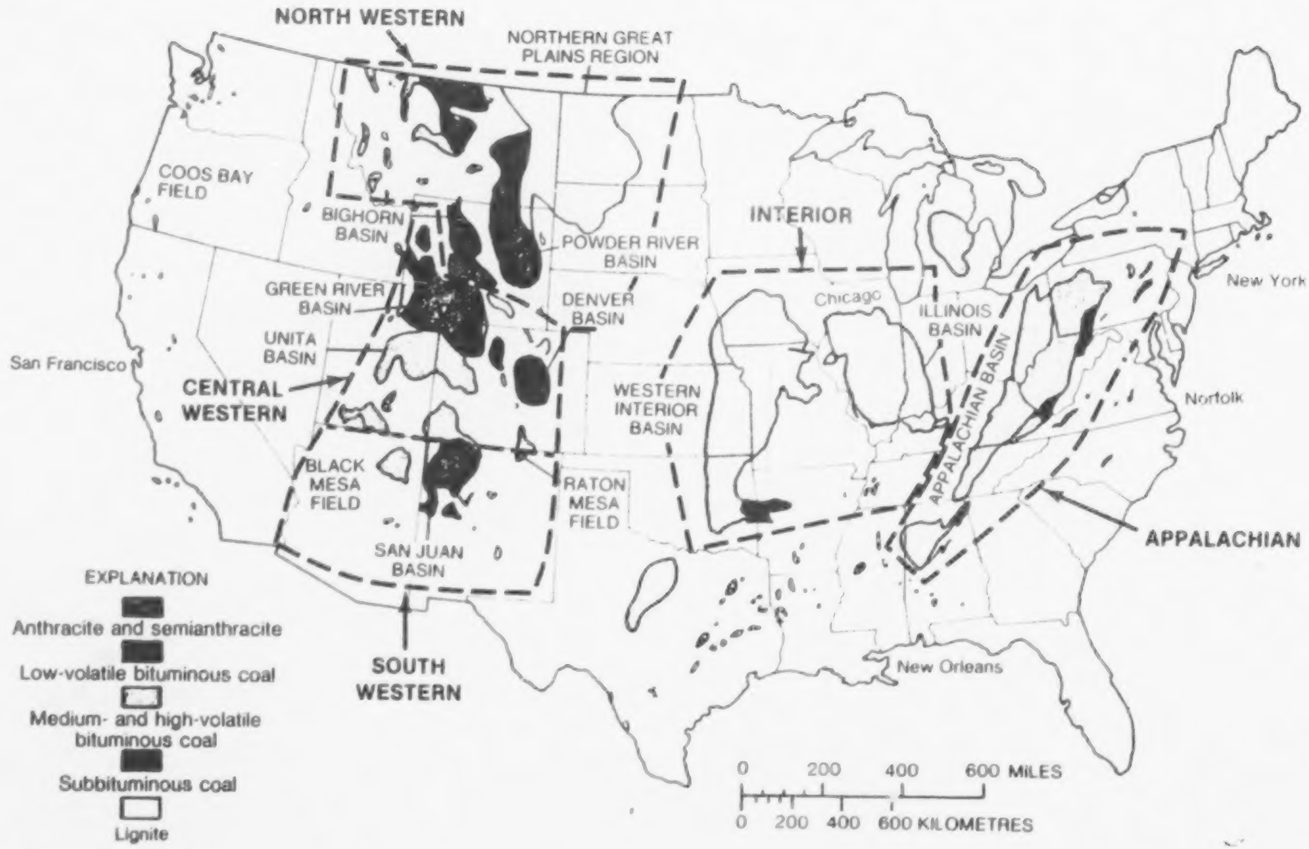
Regional coal production is largely determined by the consumption behavior of fuel users. Three factors influence the choice of coal supplies: (1) mine-mouth prices,(2) transportation costs, and (3) fuel quality (primarily the energy value and sulfur content).[1] Acid rain control measures would affect the economics of coal by significantly increasing the value of low-sulfur coal relative to high-sulfur coal, all other factors remaining equal.

Figure 1 shows the location of the nation's coal deposits. The regional character of the potential effects on the coal-market due to changes in sulfur dioxide emissions requirements stems from variations in the sulfur contents of coal among various geographical areas of the United States. Table 1 displays the distribution of each state's coal reserves by sulfur content category. A clear pattern emerges when one consults Figure 2. A large majority (74%) of coal reserves in the eastern United States are high in sulfur (greater than 2.0 pounds of sulfur dioxide per million Btu). By contrast, the majority (72%) of western reserves are low in sulfur (less than 1.2 lbs/MMBtu). The two significant eastern exceptions to this pattern are Kentucky and West Virginia. Both have significant quantities of reserves above and below 1.2 lbs/MMBtu.[2]

Table 2 analyzes each state's coal production for the utility market.[3] The first column presents the quantity of coal, in millions of tons, produced for the utility market by each state in 1980. The second column shows the percentage of that production that would not allow utilities to comply with a

FIGURE 1

U.S. COAL FIELDS


NORTH WESTERN
NORTHERN GREAT PLAINS REGION
COOS BAY FIELD
BIGHORN BASIN
POWDER RIVER BASIN
GREEN RIVER BASIN
DENVER BASIN
UNITA BASIN
San Francisco
CENTRAL WESTERN
WESTERN INTERIOR BASIN
BLACK MESA FIELD
RATON MESA FIELD
SAN JUAN BASIN
SOUTH WESTERN
INTERIOR
Chicago
ILLINOIS BASIN
APPALACHIAN BASIN
New York
Norfolk
APPALACHIAN
New Orleans
EXPLANATION
Anthracite and semianthracite
Low-volatile bituminous coal
Medium- and high-volatile bituminous coal
Subbituminous coal
Lignite
0 200 400 600 MILES
0 200 400 600 KILOMETRES

SOURCE: P. Averitt, *Coal Resources of the United States, Jan. 1, 1974*, U.S. Geological Survey Bull, 1412, at 5 (1975).

Table 1

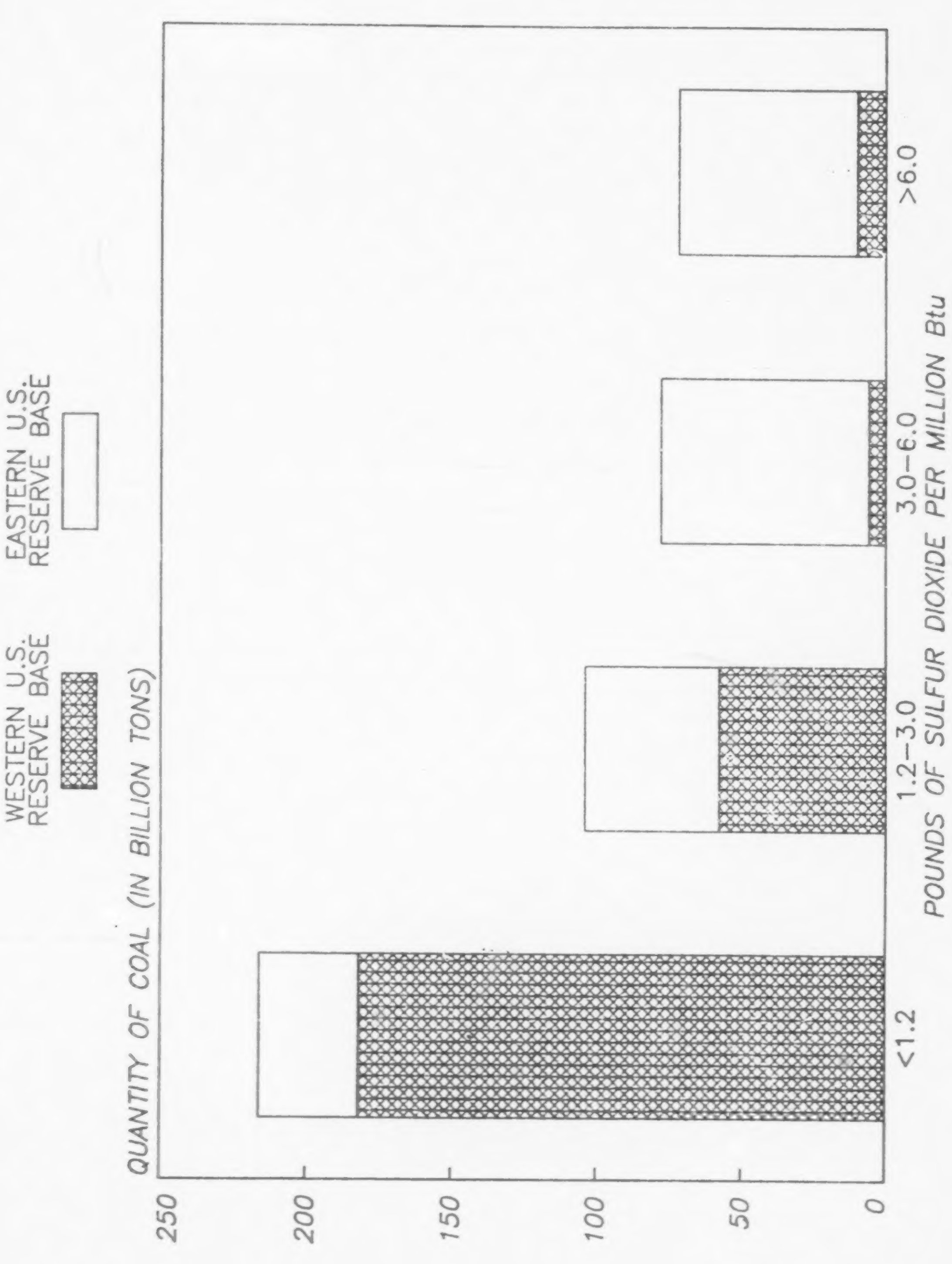
DEMONSTRATED RESERVE BASE BY SULFUR CATEGORY
QUANTITIES IN MILLIONS OF TONS
SULFUR CATEGORIES IN LBS SO2 PER MMBTU

STATE	<.9	.9-1.2	1.3-1.5	1.5-2.0	2.1-3.0	3.1-4.0	4.1-5.0	5.1-6.0	6.0	TOTAL
ALABAMA	5	1,775	1,712	1,074	1,756	438	0	0	0	6,760
GEORGIA	0	0	4	0	0	0	0	0	0	4
ILLINOIS	0	0	0	4,359	1,581	3,151	3,107	14,248	41,260	67,705
INDIANA	0	212	992	1,304	503	453	2,018	1,976	3,161	10,621
KENTUCKY	0	10,210	3,846	1,473	410	749	1,556	7,524	8,472	34,240
MARYLAND	0	1	5	296	65	175	178	107	0	826
MICHIGAN	0	0	0	0	95	0	26	0	7	128
NORTH CAROLINA	0	0	11	0	0	0	0	0	0	11
OHIO	0	0	0	0	1,119	1,294	6,684	4,429	5,509	19,035
PENNSYLVANIA	16	568	587	1,992	8,581	12,203	3,757	1,684	1,039	30,428
TENNESSEE	0	189	135	154	154	62	234	31	39	998
VIRGINIA	784	1,469	458	561	148	118	0	0	0	3,538
WEST VIRGINIA	3,407	13,724	2,932	2,791	6,029	3,084	3,535	355	4,128	39,985
EASTERN US TOTAL	4,212	28,149	10,681	14,005	20,440	21,726	21,096	30,353	63,616	214,277
ALASKA	5,805	18	332	0	0	0	0	0	0	6,155
ARIZONA	0	0	0	399	25	0	0	0	0	424
ARKANSAS	0	11	23	260	43	74	0	0	0	411
COLORADO	7,800	2,212	4,841	213	1,150	0	0	0	0	16,215
IDAHO	0	0	4	0	0	0	0	0	0	4
IOWA	0	0	0	0	0	0	0	71	2,127	2,199
KANSAS	0	0	0	0	0	190	123	210	472	995
MISSOURI	0	0	0	0	0	0	518	0	5,559	6,077
MONTANA	84,247	32,786	1,410	1	1,527	0	0	0	499	120,469
NEW MEXICO	1,277	508	2,701	7	48	0	0	0	0	4,541
NORTH DAKOTA	564	165	1,961	2,908	2,244	1,524	604	0	0	9,971
OKLAHOMA	8	750	280	26	332	144	0	21	83	1,644
OREGON	12	0	0	6	0	0	0	0	0	17
SOUTH DAKOTA	0	0	193	74	0	99	0	0	0	366
TEXAS	0	0	0	0	12,693	0	0	0	0	12,693
UTAH	399	4,733	158	0	178	1,033	0	0	0	6,502
WASHINGTON	843	16	0	578	0	143	0	0	0	1,580
WYOMING	33,527	8,560	1,007	24,214	2,606	4	0	0	95	70,014
WESTERN US TOTAL	134,482	49,761	12,911	28,685	20,845	3,212	1,246	302	8,835	260,279
UNITED STATES	138,694	77,909	23,592	42,690	41,285	24,938	22,341	30,655	72,451	474,556

Source: Adapted for OTA by E.H. Pechan & Associates from U.S. Department of Energy, Demonstrated Reserve Base of Coal in the United States on January 1, 1979, DOE/EIA-0280(79), May, 1981.

FIGURE 2

DEMONSTRATED RESERVE BASE BY SULFUR CATEGORY



OFFICE OF TECHNOLOGY ASSESSMENT, PECHAN & ASSOC., 1982

TABLE 2

COAL PRODUCTION FOR THE DOMESTIC UTILITY MARKET

	1980 Production for the Utility Market (millions of tons)	Percentage Non-Compliance*	Percentage Non-Compliance Sold on the Spot Market	Percentage Non-Compliance Exported to Other States
Alabama	15.8	82.5%	17.3%	10.1%
Arizona	10.5	0.0	0.0	0.0
Colorado	13.6	7.1	0.4	2.8
Illinois	53.4	99.9	6.6	66.0
Indiana	27.3	99.9	8.3	22.1
Iowa	0.4	95.9	14.5	0.0
Kansas	0.7	100.0	29.3	81.7
Kentucky	112.4	89.9	16.5	72.9
East	73.9	83.2	19.0	75.1
West	38.5	100.0	11.8	68.7
Maryland	1.2	100.0	22.8	24.7
Missouri	5.0	100.0	6.9	33.5
Montana	27.9	63.6	0.0	51.9
New Mexico	17.0	69.8	0.3	1.2
North Dakota	15.3	99.2	13.6	20.8
Ohio	34.3	100.0	18.9	26.5
Oklahoma	2.7	98.6	17.3	98.6
Pennsylvania	50.9	99.1	29.9	30.1
Tennessee	7.6	86.7	17.6	34.6
Texas	27.0	100.0	0.0	0.0
Utah	8.5	20.0	1.1	7.1
Virginia	13.8	87.5	10.6	71.0
West Virginia	53.1	81.3	8.1	39.4
North	30.8	97.8	7.5	46.1
South	22.3	58.5	8.8	30.1
Wyoming	89.7	16.6	0.9	9.9

* "Non-compliance" coal is defined as the amount of coal produced by a state that would not permit utilities to comply with a 1.2 pound of sulfur dioxide per million Btu emissions limit without applying control technologies.

SOURCE: From DOE/EIA Form 423 data, supplied to OTA by E.H. Pechan and Associates.

1.2 pound per million Btu sulfur dioxide emission limit without applying control technologies. The third column presents the portion of this "non-compliance" coal that is sold on the spot market, i.e., is not covered by a contract between a utility and a mining company. Non-compliance coal sold on the spot market is likely to be most vulnerable to more stringent sulfur dioxide regulations.[4] The final column shows the portion of non-compliance coal that is exported to other states -- an indicator of the potential efficacy of state-level policies designed to minimize coal market disruptions (discussed further in the final section).

III: PROJECTIONS OF CHANGES IN REGIONAL COAL PRODUCTION, EMPLOYMENT, AND ECONOMIC ACTIVITY

A. METHODS USED IN THE ANALYSIS

1. Projections of Regional Coal Production

The projections for regulation-induced changes in regional coal production presented in this report are based on a computer model modified by ICF, Inc., from the DOE National Coal Model. ICF's Coal and Electric Utilities Model (CEUM) makes it possible to compare: (1) projections of regional coal production in 1990 (and 2000) assuming that current sulfur dioxide emissions standards are maintained, and (2) projections of regional coal production in 1990 (and 2000) if significant sulfur dioxide emissions reductions, designed to control acid rain, are required. The ICF model has been used to project the effects of acid rain control proposals by: The Environmental Protection Agency, the Department of Energy, the Edison Electric Institute, and the National Wildlife Federation and National Clean Air Coalition. The model results presented here were not performed by ICF for OTA, but for the other groups mentioned above.

CEUM is a "least-cost optimization model" that chooses the combination of scrubbers, coal washing, and low-sulfur coal substitution or blending that mimimizes cost to the utility industry of achieving emissions reductions.[5] The changes in the choice of coals consumed alter regional patterns of coal production. These projected production shifts form the basis for OTA's estimates.

This report uses four ICF sulfur dioxide emissions reduction scenarios, and their attendant regional coal production scenarios, to analyze the potential magnitude of coal market impacts:

(1) Scenario I: a five million ton reduction of utility sulfur dioxide emissions by 1990, allocated according to a "regional (31-state) least cost" approach;

(2) Scenario II: a 10 million ton sulfur dioxide emissions reduction by 1990, allocated according to the formula in S.1706 (i.e., based on emissions in excess of an emissions rate of 1.2 pounds of sulfur dioxide per million Btu);

(3) Scenario III: a 13 million ton sulfur dioxide emissions reduction by 1990, allocated according to a formula similar to S.1706 and;

(4) Scenario IV: a 10 million ton reduction reduction, allocated according to a "regional least cost" approach, that shows regional coal production shifts out to the year 2000.[6]

These scenarios are used solely to illustrate the potential magnitude and direction of coal market shifts associated with the least-cost solution to the utility industry of various regulatory strategies for controlling acid rain. The model cannot predict how various acid rain proposals would, in practice, be implemented. Some analysts assert that the ICF model seriously underestimates the amount of fuel-switching that is likely to occur, while others suggest that its fuel-switching estimates are overstated.[7]

Several important factors that might influence the accuracy of ICF's projections are:

(1) Innovation in pollution control technology: The amount of fuel-switching projected to occur depends on the relative cost of that approach as compared to other emission reduction options. The ICF model attempts to reflect the best current estimates of the cost of various control options. If innovations were to significantly reduce the costs of control technology relative to fuel switching, smaller changes in regional coal production would accompany any given level of emissions reductions.

(2) Utility regulatory policy: The structure of utility rate-making may affect which emissions reduction options are most attractive to utilities. For instance, fuel adjustment provisions that allow utilities to pass increased fuel costs through to consumers without the need for a rate hearing may make fuel switching seem more attractive to utility managers than the ICF model would suggest.[8]

(3) Transportation costs: The penetration of low-sulfur western coal into eastern and midwestern utility markets is very dependent on railroad rates. A rapid escalation in western rail rates could raise the delivered price of western coal, and hence reduce the magnitude of regional shifts.

(4) Electricity growth rates: While assumptions of rate of growth in electricity demand should not significantly affect the relative amount of fuel switching as opposed to other control options, the absolute magnitude of projected increases in coal production by 1990 is highly dependent on assumptions of electricity demands.

2. Forecasting Coal Mine Employment and Economic Changes

OTA combines production projections generated by the ICF model with coal-miner productivity and income data to project the effects of acid rain controls on regional employment and economic activity. Employment in the coal mining industry is determined by two factors: the level of coal production (tons of coal) and the rate of worker productivity (tons of coal produced per miner. Historically, changes in employment have been due more to changes in productivity than to changes in production.[9] Therefore, forecasting employment requires incorporating some provision for the uncertainty of future productivity levels. This analysis presents projected coal-mine employment changes as ranges reflecting different assumptions of future productivity levels: a lower bound of a 10% decline from 1979 productivity levels and an upper bound of a 30% increase.[10] These employment projections are combined with miner income data to estimate the "direct" income impacts of acid rain controls.

Estimates of direct income effects do not reflect the full extent of the impact of changes in coal-mine employment. Economic activities dependent upon the coal-mining industry, such as rail transport and the retailers who provide household goods for coal-mine industry employees, are also affected. These indirect economic effects are likely to be quite significant in comparison to the direct effects, but are extremely difficult to estimate. Indirect effects could be realized as reductions in employment, as reductions in income to the employed, or some combination of both. In this analysis, direct income effects are combined with an income "multiplier", which provides a very rough approximation of the amount of regional economic activity that is dependent on coal mining, to derive "total" income effects of acid rain controls.[11]

C. SUMMARY OF REGIONAL ANALYSIS: PROJECTED PATTERNS OF CHANGE

Table 3 summarizes the impacts of 5, 10, and 13 million ton sulfur dioxide emissions reductions on regional coal production, employment, and economic activity. Several generalizations emerge from the analysis. First, the model projects significant growth in coal production nationwide (about 60%) between 1979 and 1990. Second, regulations designed to control acid rain, even those calling for the largest emissions reductions, do not alter the projected increases in national production. Even with the additional costs of acid rain controls, the competitiveness of coal with other fuels is not projected to suffer.

TABLE 3 SUMMARY TABLE OF REGIONAL COAL MARKET EFFECTS OF ACID RAIN CONTROL LEGISLATION

REGION	5 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION	10 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION	13 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION
NORTHERN APPALACHIA (Maryland, Northern West Virginia, Ohio and Pennsylvania)	Under current environmental regulations, annual coal production is projected to increase approximately 10 percent between 1979 and 1990; under a 5 million ton acid rain control program no change in production is projected over this period. Employment opportunities foregone are projected to range from 6,300 to 9,100 jobs (a 10 percent reduction); because there is no projected production change, employment changes from 1979 levels will be a function of productivity changes only. Direct income opportunities foregone by the regional economy are projected to be $160 to $230 million; total income opportunities lost could range from $450 to $640 million.	Under a 10 million ton acid rain control program, production is projected to decrease by about 10 percent between 1979 and 1990. Employment is projected to be 15 percent (9,800 to 14,100 jobs) less than it would have been under the 1990 base case; projected employment changes from 1979 levels range from no change to a 30 percent (23,000 job) decrease. Direct income opportunities foregone range from $250 to $360 million; total income foregone ranges from $630 to $910 million. Available projections for the year 2000 suggest these declines may only be temporary.	Under a 13 million ton acid rain control program, production is projected to decrease 13 percent between 1979 and 1990. 1990 employment levels are projected to be 20 percent (12,700 to 16,500 jobs) less than what they would have been in 1990 under current regulations, and 14 to 35 percent (10,800 to 26,100 jobs) below 1979 employment. Direct income opportunities foregone range from $330 to $470 million; total income foregone is projected to be $810 million to $1,170 million.
CENTRAL APPALACHIA (Eastern Kentucky, Southern West Virginia, Tennessee, and Virginia)	Under current environmental regulations, annual coal production is projected to increase by about 50 percent between 1979 and 1990; under a 5 million ton acid rain control program, a 62 percent increase is projected over this period. Employment is projected to be about 10 percent (8,800 to 12,700 jobs) greater than it would have been in 1990 under current regulations; projected employment changes from 1979 levels range from 25 to 80 percent (21,800 to 69,900 jobs) increase. Direct annual income opportunities created in this region range from $230 to $330 million; total annual income created ranges from $450 to $640 million.	Under a 10 million ton acid rain control program, production is projected to increase 70 percent between 1979 and 1990. 1990 employment levels are projected to be about 15 percent (15,000 to 21,700 jobs) greater than they would have been in 1990 under current regulations, and 38 to 99 percent (33,200 to 86,500 jobs) greater than 1979 employment levels. Direct income opportunities created are projected to be $390 to $560 million; total income generated ranges from $1,030 to $1,490 million.	Under a 13 million ton emission reduction, production is projected to increase 75 percent between 1979 and 1990. 1990 employment levels are projected to be 16 percent (17,300 to 25,000 jobs) greater than what they could have been in 1990 under current regulations, and 40 to 100 percent (35,800 to 90,000 jobs) above 1979 levels. Direct income opportunities created by the projected increase in coal production range from $440 to $640 million; total income generated ranges from $1,200 to $1,700 million.

TABLE 3 (continued)

REGION	5 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION	10 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION	13 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION
MIDWEST (Illinois, Indiana, and Western Kentucky)	Under current environmental regulations, annual coal production is projected to increase 31 to 34 percent between 1979 and 1990; under a 5 million ton reduction, production is projected to increase 12 percent over this period. 1990 employment levels are projected to be 16 percent (5,400 to 7,700 jobs) less than what they would have been in 1990 under current regulations, and between a 13 percent (4,200 job) decrease and a 25 percent (8,000 job) increase from 1979 levels. Direct income opportunities foregone range from $140 to $200 million; total income foregone ranges from $370 to $530 million.	Under a 10 million ton acid rain control program, production is projected to decrease 10 percent between 1979 and 1990. 1990 employment levels are projected to be 33 percent (10,500 to 15,200 jobs) below what they would have been in 1990 under current regulations, and 6 to 35 percent (1,900 to 11,300 jobs) less than 1979 employment. Direct income opportunities foregone are projected to range from $270 to $390 million; total income foregone ranges from $770 million to $1,100 million. Projections for the year 2000 suggest these declines may be reversed in the period 1990 to 2000.	Under a 13 million ton sulfur dioxide emission reduction, production is projected to decrease by 17 percent between 1979 and 1990. Employment is projected to be 38 percent (12,100 to 17,400 jobs) below what it would have been in 1990 without an acid rain control program; projected employment decreases from 1979 levels range from 13 to 40 percent (4,200 to 12,900 jobs). Direct income opportunities foregone range from $310 to $480 million; total income foregone ranges from $870 million to $1,200 million.
THE WESTERN U.S.	Under current environmental regulations, annual coal production is projected to increase approximately 138 percent between 1979 and 1990; under a 5 million ton acid rain control program, production is projected to increase 145 percent over this period. 1990 employment levels are projected to be 3 percent (1,200 to 1,700 jobs) higher than what they would have been in 1990 under curent regulations, and 80 to 160 percent (16,600 to 32,900 jobs) above 1979 levels. Direct income opportunities created by acid rain controls are projected to range from $30 to $40 million; total income benefits are projected to be $120 to $170 million.	Under a 10 million ton acid rain control program, production is projected to increase 158 percent between 1979 and 1990. 1990 employment is projected to be 13 percent (4,600 to 6,700 jobs) greater than what it would have been in 1990 in the absence of acid rain controls, and between 95 and 180 percent (19,100 to 36,300 jobs) greater than 1979 levels. Increases in direct income opportunities range from $120 to $170 million; total income increases are projected to range from $510 to $740 million.	Under a 13 million ton sulfur dioxide emission reduction, production is projected to increase 165 percent between 1979 and 1990. Acid rain controls are projected to increase employment opportunities by 20 percent (6,800 to 9,800 jobs) over what they would have been in 1990 without any change in environmental regulations. 1990 employment levels are projected to be 105 to 196 percent (21,000 to 39,300 jobs) greater than 1979 levels. Direct income opportunities created in the region are projected to range from $180 to $250 million; total income benefits range from $750 million to $1,100 million.

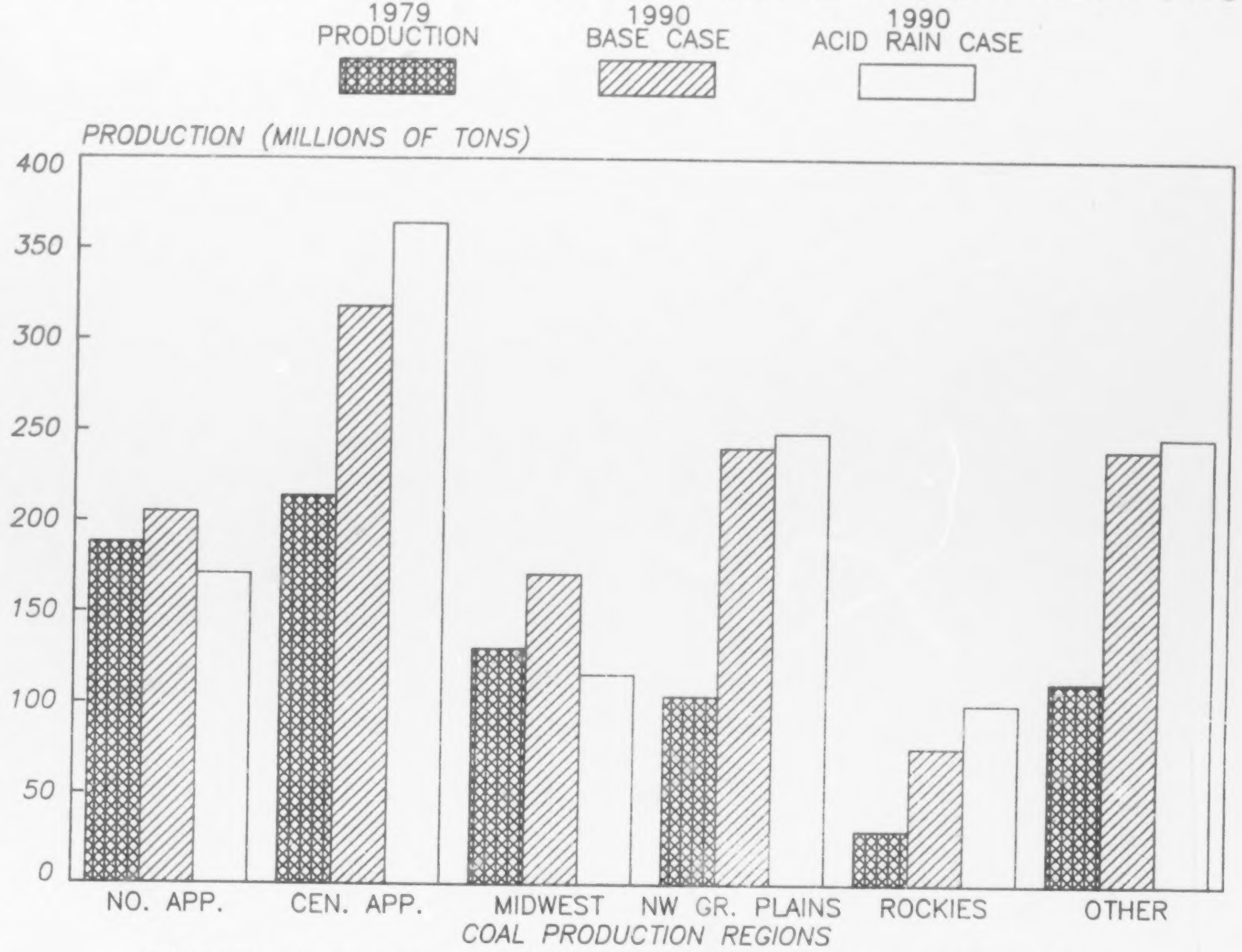
Third, even though acid rain control measures are not projected to produce changes in nationwide production, the ICF model projects that a redistribution of coal production among regions is projected to take place. Regions with low-sulfur reserves are projected to experience production growth beyond what would occur absent any change in regulations. On the other hand, between 1979 and 1990, regions with high-sulfur reserves are projected to experience production declines below what 1990 levels would be if regulations had remained the same, and -- in the case of the 10 million ton and greater reductions of sulfur dioxide emissions -- below 1979 production levels as well. The magnitude of the projected redistributions increases as sulfur dioxide emissions reductions increase, although in a less than linear fashion.[12] Figure 3 shows the projected change in regional coal production from 1979 to (a) 1990 levels assuming no change in sulfur dioxide standards; and (b) 1990 levels assuming a 10 million ton sulfur dioxide emissions reduction.[13]

The areas most likely to experience significant growth in production are Southern West Virginia, Eastern Kentucky, and the regions west of the Mississippi (particularly Colorado). The regions most likely to experience significant declines are those of Northern West Virginia, Ohio, Western Kentucky, and Illinois. However, available model projections for the year 2000 suggest that these declines below current levels may be reversed in the decade after 1990. This projected rebound in production is due to general trends in the energy market, such as growth in utility demand, and the fact that an increasing number of new plants regulated under the 1979 New Source Performance Standards will come on line after 1990.

Employment patterns are projected to correspond to changes in

FIGURE 3

REGIONAL COAL PRODUCTION: EFFECTS OF A 10 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION



SOURCE: OFFICE OF TECHNOLOGY ASSESSMENT, ADAPTED FROM ICF, INC.

production. However, the uncertainty regarding future productivity trends creates a larger band of uncertainty for employment effects than for production effects. The loss of portions of the market for high-sulfur coal could force the Midwestern and Northern Appalachian regions to forego employment opportunities on the order of 10 to 40% of what the coal-mining workforce is projected to be in 1990 in the absence of acid rain control legislation. The areas of northern West Virginia, Ohio, western Kentucky, and Illinois are projected to experience actual employment decreases below 1979 levels for emissions reductions of 10 million tons and greater. Table 4 displays projected coal-mine labor force impacts of a 10 million ton sulfur dioxide emissions reduction for coal-producing states east of the Mississippi.

These projected employment impacts would be accompanied by proportional changes in direct miner income and total monetary impacts. For _each_ of the high-sulfur coal regions of Northern Appalachia and the Midwest, these costs range from $200 million to $500 million in direct annual income losses to miners, and from $400 million to $1,200 million in total (direct plus indirect) annual monetary losses. The coal-related _benefits_ in Central Appalachia range from $200 to $600 million annually in _direct_ income gains, and _total_ annual income gains of $700 to $1,700 million. These figures, particularly in the case of total monetary impacts, must be considered very rough estimates.

Changes in coal-related employment and income may present some community-level social and economic impacts repercussions. In areas projected to experience significant declines in employment, decreases in tax revenues could cause the quantity or quality of community services to decline. In areas projected to experience rapid increases in employment, the capacity of

TABLE 4

EMPLOYMENT IMPACTS IN 1990 FOR THE EASTERN U.S. OF A 10 MILLION TON SULFUR DIOXIDE EMISSION REDUCTION

(figures in parentheses are negative numbers)

	Acid Rain Case Change From 1990 Base Case[1]		Acid Rain Case Change From 1979 Level		UMW Current Unemployment[2]
	Range[3]	%	Range	%	%
Alabama	(1000-1400)	(10)%[4]	(1100)-3000	(11)-29%	6%
Illinois	(3800-11,700)	(47)	(3300-7200)	(20-44)	10
Indiana	(1200-1800)	(20)	400-2600	9-58	23
Kentucky					
eastern	5900-8600	20	(400)-15,400	(1)-43	16
western	(1200-1800)	(15)	(1200-4300)	(11-38)	
Maryland	0	0	(100)-300	(12)-27	NA[5]
Ohio	(5700-8100)	(39)	(2400-6300)	(16-42)	46
Pennsylvania	(3100-4500)	(10)	(8100)-4400	(22)-12	18
Tennessee	(1200-1800)	(22)	800-2800	24-79	0
Virginia	2300-3300	18	(1000)-5800	(6)-36	5
West Virginia					
north	(1200-1700)	(7)	(1200-8200)	(5-34)	19
south	8100-11,600	14	33,600-62,700	105-196	

1. 1990 Base Case is what 1990 employment is projected to be in the absence of changes in environmental regulations.

2. For members of the UMW only. Figures are for April 1982.

3. The ranges represent various productivity assumptions, from a 10% decrease from 1979 levels to a 30% increase.

4. Percent change from base case is independent of productivity assumptions; therefore, no range is presented.

5. Not Available.

SOURCE: Office of Technology Assessment.

communities to provide adequate health care, housing, education, and so on, may be strained.

Comparison with other Studies

Peabody Coal and the United Mine Workers have performed similar estimates of employment and economic impacts resulting from proposed acid rain legislation. The most recent of these analyses was presented by UMW in April, 1982.[14] In that analysis, UMW stated that S.1706 (the Mitchell bill) would result in a loss of 89,000 jobs in the coal mining industry, resulting in direct economic losses of $2.4 billion and total (direct plus indirect) economic losses of $6.6 billion. These estimates measure the effects on *high sulfur coal producing areas only*.

Table 5 compares the UMW and OTA estimates. The UMW estimates assume that *all* reductions will be met by switching to lower-sulfur coal, and therefore should be considered an absolute upper bound for employment and economic losses from a 10 million ton sulfur dioxide emissions reduction. OTA assumes that the emissions reductions are met by a combination of fuel-switching and scrubber installation. In addition, whereas UMW considers "dislocations" of miners from their jobs, OTA averages job losses with job gains *within* states. OTA's estimates are between 57 and 90% lower than UMW's.[15]

Assessing the net effect of acid rain legislation on nation-wide coal production, employment, and economic activity, however, requires consideration of the offsetting increases in employment and economic activity in low-sulfur coal regions. When these are considered in aggregate, OTA estimates no significant change in employment and economic activity as a result of acid-rain induced production changes. OTA's estimates of the regional

TABLE 5

Comparison of UMW/Peabody and Office of Technology Assessment Analyses of the Employment and Economic Impacts of S. 1706 in High-Sulfur Coal-Producing Areas

	UMW/Peabody[1]	OTA Change From 1990 Base Case[2]	OTA Change From Actual 1979 Levels
Employment Losses[3]	89,000	23,000-33,000	9,000-38,000
Direct Annual Economic Losses (Millions 1981 Dollars)[4]	$2,300	$600-$800	$200-$1,000
Total Annual Economic Losses (Millions 1981 Dollars)	$6,600	$1,600-$2,300	$600-$2,700

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1. UMW/Peabody assumes all of the 10 million ton emissions reductions will be achieved by switching to lower-sulfur coal.

2. OTA assumes, based on ICF analyses, that the requisite emissions reductions will be met by a combination of fuel-switching and scrubbers. The base case assumes no change in environmental regulations.

3. These employment losses occur over the period required to implement the pollution reductions. The ranges reflect uncertainty in future productivity levels. OTA bounds employment and economic estimates by assuming that productivity might rise as much as 30% or decrease by as much as 10% between 1979 and 1990.

4. Annual monetary estimates assume implementation of emissions reductions takes 10 years, as specified in the bill, and that shifts in coal production are distributed equally over that period.

redistribution of coal-related activity, while by no means insignificant, are substantially less than the previous estimates performed by UMW and Peabody Coal.[16]

PART IV: ASSESSMENT OF POLICIES TO MITIGATE COAL MARKET DISRUPTIONS

The preceding projections of acid rain regulation-induced changes in coal production, employment, and economic activity have all been based on a computer model designed to estimate the effects of achieving the emissions reductions at the lowest cost to the electric utility industry. The "least-cost approach" projects potentially significant impacts on the coal industry in some areas. Emissions reductions designed to reduce acid rain do not, however, necessarily entail these significant regional shifts in coal market activities.

Society can choose to pursue policies aimed at mitigating disruptions in the coal market. Two forms of policies are available to minimize the effects of acid rain control measures on the coal market:

(1) The "technological approach", which involves requiring the use of control technologies by, for example, requiring that a percentage reduction of emissions be achieved regardless of the quantity of sulfur in the coal being burned; and

(2) The "local coal approach", which involves regulating the coal consumption behavior of utilities on the basis of the location of the coal supply, as in Section 125 of the Clean Air Act.

The next section assesses policies designed to mitigate potential coal market disruptions accompanying large-scale sulfur dioxide emisssions reductions. The information available to perform a comprehensive analysis of this subject is quite limited. Because of this constraint, the subject is approached in two ways. First, the one state for which some of the requisite information is available, Ohio, will be examined. Next, the potential consequences of invoking Section 125 of the Clean Air Act assuming alternative definitions of the statutory terms "significant" economic disruption and "locally or regionally" available coal supplies will be explored.

A. THE COSTS AND BENEFITS OF A TECHNOLOGICAL APPROACH TO MITIGATION

The technological approach to mitigation would require emitters to achieve their reductions by applying some circumscribed set of emissions control technologies, such as wet and dry flue-gas desulfurization units. To have some mitigative effect, the policy must minimize the potential cost advantage of switching to a lower-sulfur coal.

As presented earlier, Scenario II (S.1706) is projected to result in an employment decrease, for the state of Ohio, of between 5600 and 8100 workers from the base case. OTA estimates that if fuel-switching were prohibited in Ohio, the annual costs of sulfur dioxide emissions reductions would increase by as much as $136 million.[17] The benefits of this prohibition can be estimated by calculating the economic value of avoiding coal-industry employment losses.

To simplify the analysis, assume that a technology-based Federal regulation would totally eliminate fuel-switching from non-compliance coal. It follows that a technology-based state regulation imposed by Ohio would eliminate fuel-switching for, at most, that percentage of Ohio non-compliance coal consumed in Ohio, approximately 73% of Ohio's production for the utility market (as shown in Table 2).

If the state-imposed technology standard could mitigate 73% of the employment losses projected to occur from the base case, the direct economic benefits of saving these jobs would range from $105 to $155 million per year, an amount about equal to the additional control costs incurred. A federally-imposed, technology-based regulation would provide even greater benefits for Ohio, but other states would then also incur increased control costs without gaining coal-related benefits.

B. ASSESSMENT OF SECTION 125 AS POLICY FOR THE MITIGATION OF COAL MARKET DISRUPTIONS

The second major approach available to policy-makers for mitigating the coal-market effects of proposed emissions reductions does not directly address emissions control techniques, but focuses instead on regulating the coal consumption of emitters based on the location of the coal supplier. Section 125 of the Clean Air Act allows states or EPA to prohibit the use of coals that are not produced "locally or regionally" if such an action would "prevent or minimize significant local or regional economic disruption or unemployment".[18] No ruling has yet been made under Section 125, although EPA has proposed a ruling based on a petition filed by the United Mine Workers and others in the state of Ohio in 1978.[19]

The potential significance of the Section 125 approach to mitigating coal market shifts is difficult to judge because neither the statute, nor EPA in its ongoing proceedings, have defined "significant" economic disruption or "locally or regionally" available coal.

1. The Consequences of Alternative Definitions of "Significant" Economic Disruption or Unemployment.

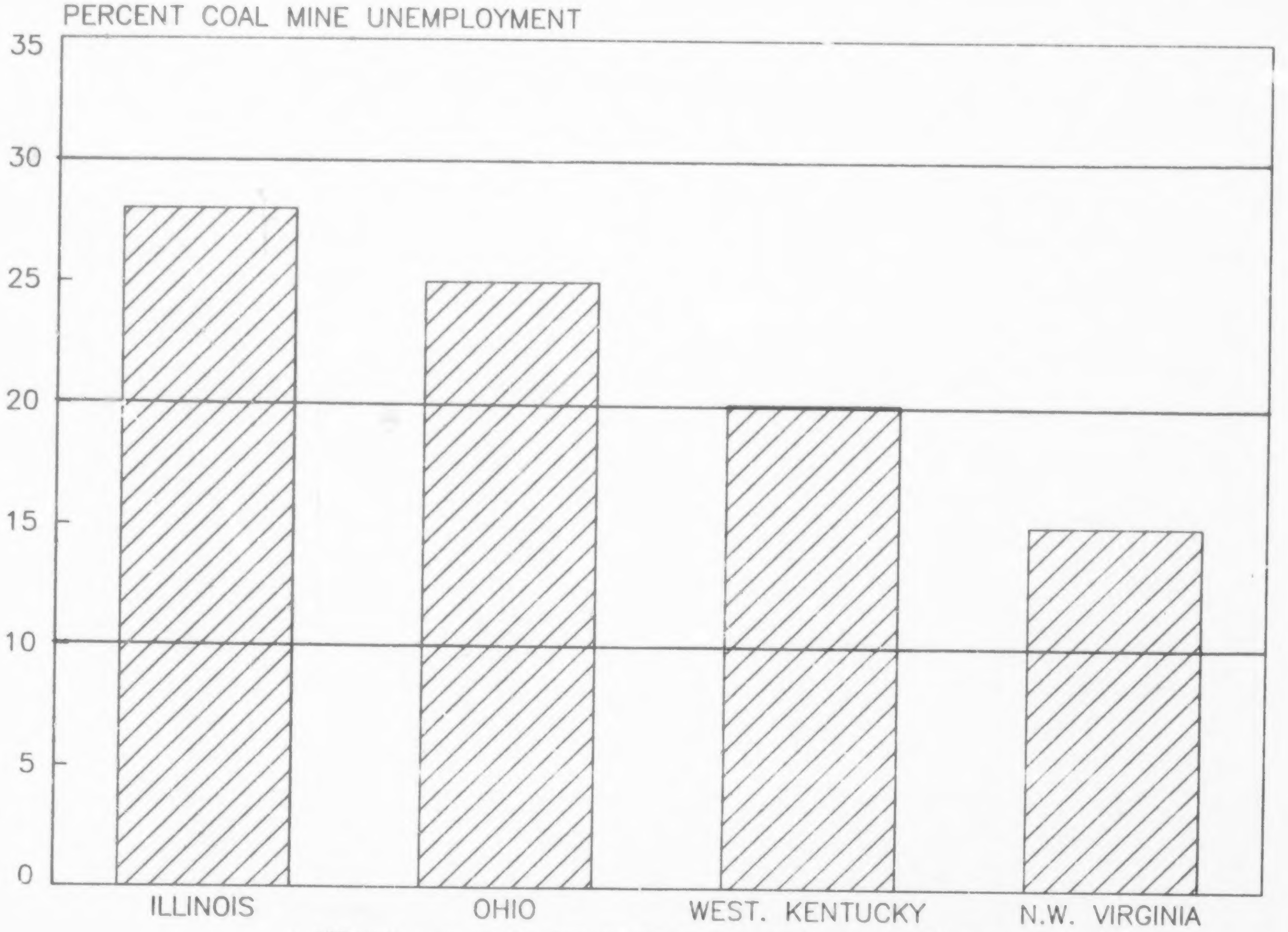
The case record provides little guidance as to how Section 125 might be applied if acid rain legislation were to be enacted.[20] However, it is possible to consider how extensively Section 125 might be invoked if acid rain control legislation were implemented. One can define alternative hypothetical "thresholds of significance", i.e., the level of impact judged by EPA to warrant the use of Section 125, and then analyze how many states these various definitions could apply to. Figure 4 shows states and regions of states to which Section 125 could be applied if the "threshold of significance" were alternatively defined as 10%, 20%, and 30% reductions in coal mine employment. The employment figures are based on emission reductions projected to occur under Scenario II -- a 10 million ton sulfur dioxide emissions reduction, allocated according to the formula of S.1706.

As the figure shows, four areas would be threatened by coal employment losses of greater than 10% from 1979 levels: Ohio, Illinois, northern West Virginia, and western Kentucky.[21] Northern West Virginia is projected to experience a 15% (3600 job) decrease in coal-mining employment. Western Kentucky is projected to lose approximately 20% of its coal-mining employment, or about 2300 jobs. Ohio is projected to lose approximately 25% of its employment in coal mining, while for Illinois the percentage is slightly larger at 28%. These percentages represent approximately 3600 lost jobs in Ohio and 4600 in Illinois. There are no states for which coal-mining employment is projected to decrease by more than 30% below 1979 levels, under assumptions of constant productivity.

It is unclear whether the EPA Administrator would judge the impacts to be above the so-called "threshold of significance"; they do, however, meet the criterion of jeopardizing "the jobs of thousands of miners" cited in the rationale for the "reproposed" determination in the Ohio case.[22]

FIGURE 4

STATES PROJECTED TO BE ELIGIBLE* FOR ACTION UNDER SECTION 125 GIVEN ALTERNATIVE "THRESHOLDS OF SIGNIFICANCE"



* ASSUMING A 10 MILLION TON SULFUR DIOXIDE REDUCTION AND THAT COAL MINE EMPLOYMENT IS THE CRITERIA FOR SIGNIFICANCE. SOURCE: OTA

2. Consequences of Alternative Definitions of "Local or Regional" in Section 125

Two considerations influence the relationship between definitions of "local or regional" and the goal of maximizing coal market stability in the face of requirements for large-scale sulfur dioxide emissions reductions. The first is the eventual destination of a state's coal production. If, for example, "local or regional" were to be defined as being synonymous with state boundaries,[23] invoking Section 125 would leave a state's high-sulfur coal production that is exported to other states vulnerable to regulation-induced disruption. Table 2 presents high-sulfur exports as a percentage of high-sulfur production for each coal state. As much as 66% of Illinois' and 82% of Kentucky's high-sulfur coal would remain vulnerable under such a definition.

The amount of coal protected can only be increased by expanding the area considered local or regional in order to incorporate larger percentages of states' high-sulfur coal production. For example, if the Illinois "region" were expanded to include Missouri, Indiana, and Michigan, the amount of non-compliance coal exported outside the "region" would fall from 66% (when considering only Illinois) to 21% (when considering all four states). For Pennsylvania, if the adjacent states of Ohio, Maryland, New York, New Jersey, Delaware, and Maryland were included in its region, the proportion of vulnerable coal -- i.e., coal that would not be protected under potential Section 125 actions -- would fall from 30% to 4%. Similarly, if Ohio's region were expanded to include Michigan, Indiana, and Pennsylvania, its amount of "vulnerable" coal would fall from 26% to 10%.

The second factor relevant to designing appropriate regional boundaries is the homogeneity of the characteristics of coal reserves. If reserves of vastly different sulfur contents were included in the same region, disruption of coal markets would probably not be minimized. For instance, defining a region that included Southern West Virginia and Ohio would still permit dramatic shifts from high- to low-sulfur coal within the statutory constraints of Section 125. The quantity of "compliance" reserves in southern West Virginia is equivalent to 90% of the entire Ohio reserve base. On the other hand, Ohio and Northern West Virginia coals are sufficiently similar in sulfur content so that one area's coal would probably not be significantly more attractive to regional buyers than another's.

Thus, Section 125 could effectively mitigate coal-market shifts resulting from proposed emissions controls only if regions were designed to include a significant portion of a state's coal customers, while excluding large reserves of coal with differing sulfur contents. In practice, a balance would have to be struck between defining regions large enough to protect sufficient amounts of coal, and small enough to exclude coal reserves of significantly different qualities.

The principal goal of the Clean Air Act is environmental protection. Section 125 is somewhat unusual in that its primary purpose is to mitigate economic impacts. It protects one economic sector (the high-sulfur coal industry) from actions taken by another sector (the utility industry) to comply with environmental standards.[24] As the analyses in sections 1 and 2 above suggest, definitional issues may present significant problems in implementing Section 125. Any decisions restricting competition among coal suppliers or defining what levels of unemployment warrant federal action are liable to evoke a great deal of political controversy. Indeed, EPA concludes that its "experience with Section 125 casts considerable doubt on the workability of this portion of the statute". EPA states that the process has proved to be cumbersome and conflict-creating.[25]

3. Other Mitigation Proposals

EPA's proposed determination under Section 125 states that "there should be a choice besides simply allowing the disruption to occur or interfering with the workings of a particular market to reduce an economic and employment impact". EPA presents a third alternative, analagous to programs providing adjustment assistance to workers, firms, and communities that have been injured by foreign competition because of free trade laws.[26] A policy could be designed to allow special compensation to workers and communities if they are seriously affected by environmental regulations. Workers could be given special retraining to help them find jobs in other industries in their communities. Or, if alternative employment opportunities are relatively unavailable in the areas in which workers now reside, payments could be provided to facilitate their movement to other areas where opportunities may be available.

4. Conclusions: Distributional Effects and Trade-offs

The net national effect of acid rain control measures on the coal production, employment, and economic activity is projected to be minimal: the economic opportunities created in low-sulfur coal producing areas tend to balance out the economic opportunities eliminated in high-sulfur areas. The coal-related impacts are thus almost entirely distributional in character. Policies are available, as discussed in Part IV, to moderate the coal market shifts threatened by acid rain control legislation. These policies, however, may be difficult to implement and would increase the costs of achieving the emissions reductions entailed by the control programs. Such costs would primarily take the form of further increases in the price of electricity (and the goods dependent thereon) as one departs from strategies designed to minimize emissions reduction costs to the utility industry.

The "coal issue" does not necessarily require policy choices between the vitality of the high-sulfur coal industry and reducing sulfur dioxide emissions. It is possible to have both. Society can significantly reduce sulfur dioxide emissions and protect the high-sulfur coal industry, but only by imposing costs on electricity consumers and constraining the market potential of the low-sulfur coal industry. Whether these additional costs are warranted by the regional economic and employment benefits that may result is a significant issue confronting the Congress in the Clean Air Act reauthorization of 1982.

FOOTNOTES FOR THE COAL ANALYSIS

1. For a discussion of the factors affecting the choice of coal, see Congressional Budget Office, The Utility Industry, the Coal Market, and The Clean Air Act, April, 1982; and Martin Zimmerman, The U.S. Coal Industry: The Economics of Public Choice, (Cambridge, MA: M.I.T. Press, 1981).

2. Kentucky and West Virginia account for 85% of the low-sulfur coal reserves east of the Mississippi. To place these figures in perspective, the low-sulfur coal reserves east of the Mississippi could sustain the entire level of United States coal production for a period of approximately 4 years (assuming 1979 coal production of 778 million tons, a growth rate of 3% per year, and 50% recoverability of reserves).

3. This data is derived from DOE Form 423, and has been supplied to OTA by E.H. Pechan and Associates.

4. The degree to which contracts constrain fuel-switching is a major uncertainty in this analysis. 88.5% of coal purchased in 1980 was purchased on contract. See D.O.E., Cost and Quality of Fuels for Electric Utility Plants: 1980 Annual, DOE/EIA-0191(80), p. 10.

   It is not clear whether changes in environmental regulations would absolve purchasers of their contractual obligations. A contract can contain provisions (such as "force majeure" clauses) which directly address the obligations of parties in the event of changes in environmental regulations. Buyer or seller could be relieved of all or part of the contractual obligations. See Scott, "Coal Supply Agreements", 23 Rocky Mountain Law Institute 107 (1979). In the absence of contractual language specifically dealing with these circumstances, a court could still allow a utility to avoid contractual responsibilities. It could interpret the language of the contract, or it could use common law doctrines and frame equitable relief which relieves the buyer of liability. Examples of common law principles available to the court are: mistake, impracticability, and frustration of purpose. See Calamari and Perillo, Contracts Ch. 13 (1977).

   Consolidation Coal Company has stated that it has one contract with the Lansing Board of Water and Light for 500,000 tons per year, broken because of state implementation plan requirements. An occurence more frequent than contract cancellation is simply the refusal to renew contracts when they expire. Personal communication with Daniel E. Rogers, of Consolidation Coal, April 14, 1982.

5. For a description of ICF's Coal and Electric Utilities Model see the executive summary in ICF, Inc., Capabilities and Experience in the Coal and Electric Utility Industries, May 1981.

6. For further description of these scenarios, see Appendix A of the OTA staff paper The Coal Question: The Impacts of Large-Scale Sulfur Dioxide Emissions on Regional Coal Production, Employment, and Economic Activity, forthcoming, June 1982. Scenario I was performed by ICF for ANL, DOE, and EPA. See Alternative Strategies for Reducing Utility SO_2 and NO_x Emissions, Draft Report, September 1981. Scenarios II and III were

performed by ICF for the Edison Electric Institute. See the EEI memo dated February 8, 1982 on ICF analysis of the Mitchell Bill (S.1706). Scenario IV was performed by ICF for the National Wildlife Federation and National Clean Air Coalition. See Cost and Coal Production Effects of Reducing Electric Utility Sulfur Dioxide Emissions, November 14, 1981.

7. For analyses asserting that ICF underestimates fuel switching, see the testimony of Chris Farrand of Peabody Coal Company on S.1706 to the Committee on Environment and Public Works, U.S. Senate, October 29, 1981. If this is the case, the projections presented here have underestimated the magnitude of regional redistribution of production.

   On the other hand, utility analyses of how the Mitchell bill would be implemented for their systems show little evidence of potential coal market disruptions. An analysis by American Electric Power (the nation's largest utility and coal consumer), states that only 4 of their 27 units would switch to lower sulfur fuels, while the others would retrofit scrubbers or be retired. See AEP, "Economic Impact Summary of Mitchell Bill (S.1706) on Customers of the AEP System", February 23, 1982. It should be noted that the ICF model assumes that no policies for minimizing the disruptions in the coal market are implemented (see Part IV).

8. Most public utility commissions do not allow utilities to earn a return on capital investment (such as a scrubber) until it becomes operational. Thus, because changes in fuel prices can be passed on immediately to consumers, utilities might prefer fuel switching. This could be counteracted somewhat by provisions allowing utilities to earn cash returns on "construction work in progress" before the equipment becomes operational.

9. Over the period 1960 to 1970, coal-mine employment declined from 190,000 to 144,000, a 3% average annual decrease. Over the same period, coal production increased from 434 million to 613 million tons, an average annual increase of 4%. The discrepancy between employment and production trends is accounted for by increases in average coal-miner productivity during this period. The average coal production per miner increased from about 12 to about 18.5 tons per day -- an average annual increase of nearly 4%. The increasing proportion of surface mining -- from 32% of annual production (or 139 million tons) in 1960, to 44% (or 270 million tons) in 1970 -- accounts for some of the gains, since the average surface miner produces about 3 times as much coal per day as the average underground miner. In addition, surface miner productivity increased at an annual rate of 4.5% during this period, while underground miner productivity increased at a 2.6% annual rate. This productivity increase made it possible for production to increase with decreased employment. Data are from D.O.E., 1980 Annual Report to Congress: Volume 2, DOE/EIA-0173(80)/2; and The President's Commission on Coal, Coal Data Book, Washington, D.C., February 1980.

10. These productivity ranges have been chosen somewhat arbitrarily. The 30% increase in productivity was chosen as one bound because it reflects the historical maximum. The 10% decrease was chosen to account for factors that may contribute to decreased productivity, such as more stringent mine health and safety or surface reclamation regulations, or a shift from surface to underground mining. After a steady decline over the past 12

years, productivity is finally beginning to rise again, but future trends are uncertain. For a discussion of factors affecting worker productivity in coal mining see Office of Technology Assessment, Direct Use of Coal, April 1979, Chapter IV. See also Electric Power Research Institute, The Labor Outlook for the Bituminous Coal Mining Industry, EPRI EA-1477, Final Report, August 1980.

11. To arrive at employment estimates, OTA divides projected production increases by state-by-state productivity data developed from DOE data. These employment changes are then combined with miner salary data to arrive at direct income effects. Miner income was assumed to be $25,700 per year. This figure was taken from United Mine Workers of America, "Economic and Employment Impacts of Proposed Acid Rain Legislation", April 22, 1982. Indirect economic impacts were estimated using an "economic base technique". For each state, multipliers were developed by calculating the ratio of income derived from the "base" or "primary" sector to income derived from the "service" or "secondary sector". For furthur detail on the methodology used, see Appendix B of the OTA staff paper, supra footnote 6.

12. As emissions reduction requirements increase beyond a certain level, the proportion of reductions that are projected to be met by "fuel-switching" decreases relative to the proportion met by scrubbers.

13. Of the 90 million tons of annual production lost by the Eastern and midwestern high-sulfur coal market by 1990, about 50 million tons is gained by eastern low-sulfur producers and 40 million tons by western producers.

14. United Mine Workers of America, "Economic and Employment Impacts of Proposed Acid Rain Legislation", April 22, 1982.

15. Most of the difference can be attributed to different production change projections. On a national basis, UMW's projections of production losses to high-sulfur coal areas are approximately twice those of OTA. This difference overwhelms differences in productivity and multiplier data.

16. The Department of Energy has also performed some rough calculations on the coal-market effects of sulfur dioxide emissions reductions. DOE estimates that between 5100 and 6400 mine workers will be affected for every million tons of sulfur dioxide emissions reduction achieved through fuel-switching. Using the UMW assumption that all reductions are achieved through fuel-switching, DOE estimates between 51,000 and 64,000 jobs lost due to a 10 million ton emission reduction. See Department of Energy, Costs to Reduce Sulfur Dioxide Emissions, DOE/PE-0042, March 1982, p. 13 of appendix.

17. E.H. Pechan and Associates, Inc., Reducing Sulfur Dioxide Emissions for the Electric Utility Industry, prepared for the Office of Technology Assesment, p. 144. This is an OTA contractor report which will be available June, 1982.

18. Clean Air Act Section 125, 42 USC § 4725 (a) as modified by the National Energy Policy and Conservation Act of 1978, P.L. No. 96-619, Section 107,

92 STAT. 3286, 42 USC § 3286. Section 125 was modified by NEPCA to require that no ruling under Section 125 could "subject the consumer to significantly higher energy costs", a provision that further complicates the implementation of the Section.

19. 43 Fed. Reg. 60,654-55 (1978); 44 Fed. Reg. 52,031-32 (1979); 48 Fed. Reg. 8106-8109 (1981).

20. EPA has examined one case in which it was petitioned to invoke Section 125 by the United Mine Workers in Ohio. On December 28, 1978, EPA proposed to invoke Section 125 to prohibit several Ohio utilities from switching to coal that was not "locally or regionally available", based on the finding that utility actions to achieve SIP compliance in Ohio would decrease employment in the Ohio coal-mining industry by 35%. 43 Fed. Reg. 60,654-55 (1978). After public hearings and further investigation, EPA revised its estimate of employment losses to 6%. Based on these revised estimates, EPA proposed, on September 6, 1979, that these employment impacts "are not sufficiently significant to necessitate further action" under Section 125. 44 Fed. Reg. 52,031-2 (1979). Later, on January 26, 1981, EPA again revised its estimates, this time upward, to a 12.4% reduction in the mining labor force. It maintained that these impacts were below the "threshold of signficance" required to trigger the use of Section 125. 48 Fed. Reg. 8106-9 (1981). EPA has yet to make a final ruling on this case. For further detail on this case, see the OTA staff paper, supra footnote 6.

21. This assumes constant productivity. While unemployment in more states would be greater if productivity increased, those impacts are not the result of changes in the Clean Air Act -- a condition required to trigger the use of Section 125.

22. 44 Fed. Reg. 52,031-32 (1979).

23. While it has presented no generalizeable ruling on the matter, in the Ohio case EPA does limit the area of consideration to the state. In its second reproposed determination of January 1981, EPA states:

    > Ohio coal curtailments resulting from switches by out-of-state utilities, however, are not factored into the economic and employment impact estimates. EPA considers these curtailments to be outside the scope of the analysis for the Ohio Section 125 proceedings since the southeastern Ohio coal producing areas is not locally or regionally situated with respect to these out-of-state major fuel-burning stationary sources. 46 Fed. Reg. 8108.

    EPA cites no rationale for excluding out-of-state utilities beyond this statement. It should be noted that the Ohio coal fields lie adjacent to Pennsylvania, West Virginia, and Kentucky, and thus is some cases defining local or regional as the state may exclude some major fuel-burning sources that are closer in proximity to the coal mine than some major fuel-burning sources in the state of Ohio.

24. For an analysis of the non-environmental consequences of the Clean Air Act Amendments of 1977, see Peter Navarro, "The Politics of Air Pollution",

The Public Interest, Spring 1980. See also Bruce Ackerman and William Hassler, Clean Coal/Dirty Air, (New Haven, CN: Yale University Press, 1981).

25. 46 Fed. Reg. 8109.

26. See the Trade Act of 1974. 19 USC Sections 2272-2293. The rationale for this provision is described in the following extract from the legislative history (P.L. 93-618):

> In the Trade Expansion Act of 1962, Congress established a special program of worker adjustment assistance in the belief that the special nature of employment dislocation resulting from changes in trade policy necessitated a level of worker protection somewhat beyond what is available through regular State unemployment insurance programs. Increases in imports of a given type of product can simultaneously and abruptly produce a combination of situations which have a particulary severe effect on employment. Because entire industries and not only individual firms may be adversely affected, workers may not have any realistic opportunity to find new employment which is at all related to the skills and training they may have accumulated over many years. Moreover, it can happen that the affected industry is one which is concentrated in a particular region with the result that the possibility of quickly absorbing displaced workers into others types of employment available in the area would be minimal. Thus, because trade-related unemployment may differ somewhat in nature from unemployment arising from other causes and because such trade-caused unemployment is a result of a Federal policy of encouraging increased foreign trade, worker adjustment assistance was provided for in the 1962 legislation.

For information on trade adjustment assistance programs, see Congressional Research Service, Trade Adjustment Assistance for Workers: Program Growth and Changes, Issue Brief Number IB80082, originated 8/22/80, update 12/8/81. For more general information on special jobless benefit programs, see National Commission on Unemployment Compensation, Unemployment Compensation, Final Report, July 1980, chapter 9.

Appendix FF

1. THE EFFECTS OF OZONE ON AGRICULTURAL PRODUCTIVITY[1]

INTRODUCTION

Ozone has been recognized as a factor in crop production for over two decades (Heck et al. 1977). Ozone (O_3) alone or in combination with sulfur dioxide (SO_2) and nitrogen dioxide (NO_2) is responsible for up to 90% of the crop losses in the United States caused by gaseous air pollutants (Heck et al. 1980). Previous estimates of crop losses, using limited available data and assuming that all areas of the United States just met the current O_3 standard (0.12 ppm for 1 h), showed a loss of one to two billion dollars (2-4% of crop production) (Heagle and Heck 1980). These efforts were limited by lack of available field-generated data on crop loss, limited spatial and temporal resolution of ozone data, and an inability to integrate these factors with crop distribution and productivity data. The present analysis utilized data recently developed by the National Crop Loss Assessment Network (NCLAN) (Heck et al. 1982) in combination with more recent crop and ozone data to more accurately determine impacts on the productivity of corn, wheat, soybean, and peanut crops.

Ozone impacts on crop productivity were estimated by using new NCLAN dose-response relationships, 1978 ozone monitoring data, and 1978 agricultural statistics. The selected crops represent a range of ozone sensitivity from sensitve (peanut) to sensitive/intermediate (soybean) to intermediate (wheat) to tolerant (corn). Of the major United States commodities in the ten USDA farm-production regions (Table 1), these four crops represent 62% of the acres harvested and 63.5% of the dollar value. The dose-response functions are linear equations utilizing results from open-top field chamber studies.

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[1] Research sponsored by Interagency Agreement OTA No. 233-2420.0 and DOE No. 40-1159-81 for the Office of Health and Environmental Research, U.S. Department of Energy, under contract W-7405-eng-26 with Union Carbide Corporation. Publication No. 1996, Environmental Sciences Division, ORNL.

The equations predict percent-yield reduction for given ozone exposures represented as the maximum 7-h/d seasonal mean in parts per billion (ppb) with zero reduction at the background concentration of 25 ppb. Hourly ozone measurements from approximately 300 selected monitoring stations were used to estimate non-urban concentrations in individual counties for use with the county agricultural data. Maximum 7-h/d ozone data were used to estimate monthly means which were used to calculate growing season means. The 1978 crop yields represent potential yields that have been reduced due to current levels of atmospheric pollutants including ozone. Therefore the dose-response equations were used to estimate potential gains in productivity which could be achieved if ozone levels were reduced to 25 ppb. The dose-response equations were used to combine county-level ozone and productivity data for each crop to estimate yield gains. County-level maps portray the patterns of ozone impacts while summary tables define the magnitude of the impacts.

UNCERTAINTY ASSOCIATED WITH THE ANALYSIS

Any attempt to analyze regional impacts of some perturbation on the growth, yield, or quality of a natural resource is difficult because of the complex interacting factors and the constantly changing variability of natural systems across regions as broad as the agricultural land area of the United States. Therefore, this analysis is an attempt, based on the best data available, to understand certain general patterns that can be extracted from the integration of data bases associated with greatly different levels of

Table 1
1978 TOTAL PRODUCTION AND CASH VALUE OF MAJOR UNITED STATES CROPS (USDA 1980)

Crop	10^6 acres harvested	Value, $ x 10^9
Corn for grain	70	15.9
Soybeans	63	12.5
Hay	62	6.6
Wheat	57	5.3
Cotton	12	3.0
Tobacco	<1	2.7
Grain sorghum	14	1.9
Potatoes	1	1.2
Rice	3	1.1
Barley	9	0.9
Tomatoes	<1	0.8
Peanuts	2	0.8
Oats	11	0.7
Sugar beets	1	0.6
Dry beans	2	0.3
Rye	1	0.05

uncertainty. The following caveats are given to aid in interpreting the results.

1. Summary data are expressed as yield gains expected by reducing ozone concentrations to background levels (25 ppb), rather than as simple loss percentages. To estimate gains, it was necessary to assume that the data in the 1978 Census of Agriculture reflects a percentage loss due to ozone.
2. The dose-response functions are in most cases developed from the response of one or two cultivars of the crop species (except for wheat, with data for four cultivars). Cultivars are known to vary within a species in their sensitivity to air pollution. For the purposes of this analysis it was necessary to assume that the data were representative of the average response of all cultivars of that species that are commercially grown.
3. The dose-response functions are derived from data for one or two growing seasons. It is necessary to assume that climatic factors influencing plant response to ozone in those years were representative of plant response over long-term average climatic conditions.
4. Crops for which dose-response curves were generated were maintained under adequate moisture conditions at all times. Plants under periodic moisture stress, such as that which frequently occurs in field crops, will respond differently to ozone than will plants adequately supplied with moisture. However not enough information is available on the role of soil moisture in determining field response of plants to ozone to be able to modify the estimates to take this factor into account.
5. Procedures used to interpolate ozone data may not have effectively eliminated all urban influences on the interpolated values for rural counties.
6. Because measured crop loss due to ozone as determined by NCLAN studies cannot separate potential contributions to that loss due to ambient levels of acid precipitation, the dose-response functions include any contribution of ambient acid precipitation. Interactions with SO_2 and NO_x in the test chambers were excluded through the filtering process; however, SO_2 and NO_x could locally influence O_3 impacts.
7. The calculation of economic impacts by simple multiplication of yield gains by price does not take into account such market considerations as price effect. Thus, while dollar gains may be overestimated by this method, the yield changes, in kilograms or bushels, provide a starting point for more sophisticated econometric analysis.

It is not currently possible to place an aggregate quantitative estimate of uncertainty associated with figures for ozone impacts on these selected crops. The present analysis is a best estimate, given the limitations defined, with the anticipation that research hypotheses generated by the analysis will, through an iterative process of research and reassessment, lead to more accurate analyses in the future.

SOURCES OF DATA

Crop Response Data

The data used were obtained from the 1980 National Crop Loss Assessment Network (NCLAN) Annual Report (RMC 1981) or from the earlier work of groups using the methods later adopted for the NCLAN project (Heagle and Heck 1980, Heagle et al. 1979a,b fall into the latter category). Published literature from 1976 to 1980, was scanned before choosing the NCLAN data set. A 1976 cutoff was selected because the review of Heck et al. (1977) was completed in that year. Thirty-four articles were abstracted but several serious problems were common to most of the published data which precluded using these sources to develop dose-response functions for regional asessments of ozone impacts. Inconsistencies and deficiencies included the following:

1. Foliar injury was the usual response measured, and a single acute ozone dose was normally applied (at most 2 or 3 doses). Studies involving both foliar injury and yield often showed a poor correlation between the two responses.

2. Methods for exposure varied widely from greenhouse chambers to a variety of field enclosures.

3. Many did not give adequate information on O_3 concentration or yield data to permit construction of a dose-response curve; given the variability in methods, assembling of such a data set from several sources was not attempted.

4. Report of O_3 dose was not consistent. Comparisons, for example, between doses expressed as hours >0.10 ppm and as a maximum 7-hd seasonal mean are not possible.

The NCLAN data provided a set of consistent experimental results. All of the O_3 dose-yield response data were obtained from plants grown in open-top field exposure chambers of the same design. Yield (economic) data for all crops were available as grams per plant. Ozone dose was expressed as a maximum 7-h/d seasonal mean (this method precludes a direct discussion of acute effects on yield). Enough data were available for each crop at different O_3 concentrations to approximate a dose-response curve. In summary, all of the yield-loss functions are based on ozone expressed as a maximum 7-h/d seasonal mean and the corresponding crop yield in terms of grams per plant. Yield data from plants receiving charcoal-filtered air (25 ppb O_3) are considered as controls.

1. Soybeans -- Three sets of soybean data were available, two on cultivar Davis (1977 and 1978) and one on Corsoy. The two sets of Davis data report yield as percent of charcoal-filtered control yields (Heagle and Heck 1980). The actual yield data, two chambers per treatment, are found in the 1977 and 1978 Field Reports of the USDA Air Quality Research Group (Raleigh). The Corsoy data were obtained from the section by Kress and Miller in the first annual NCLAN report. These authors supplied the individual chamber averages comprising the treatment means shown in the report (4 chambers/treatment). The Corsoy yield data were corrected by the authors for the effects of a virus infection. Details are given in the NCLAN report (RMC 1981). Ozone concentrations were reported as two sets, one as a 7-h/d seasonal mean for the exposure period only, August 6 to September 30. The latter set of concentrations was used because it gave a more conservative loss estimate.

2. Winter Wheat -- Wheat data were taken from Table 2 of Heagle et al. (1979b) for plants grown in the ground and the 1977 Field Report of the USDA Air Quality Research Group (Raleigh). Data were available for four cultivars (4 chambers per treatment): Blue boy II, Coker 47-27, Holly, and Oasis.

3. Corn -- Yield data on cultivar Coker 16 were taken from Table 2 of Heagle et al. (1979a); individual chamber yields (5 chambers per treatment) were obtained directly from A. S. Heagle.

4. Peanuts -- Yield data of cultivar NC-6 were taken from the NCLAN report (RMC 1981); the individual chamber averages (4/treatment) were obtained directly from A. S. Heagle.

Crop Data

The 1978 Census of Agriculture provided county-level yield statistics for the crops. The Census of Agriculture, conducted by the Department of Commerce, is an extensive national inventory based on responses of mail questionnaires which is conducted approximately every five years. County estimates are adjusted for nonrespondants. Wheat yields represent winter and spring wheat combined. Corn yields for grain were used in the analysis.

Ozone Data

The estimation of seasonal maximum 7-h/d means required station selection, temporal averaging, and spatial interpolation. The analysis was performed by U.S. Environmental Protection Agency, Research Triangle Park, North Carolina (J. A. Reagan, personal communication). Approximately 300 of a total of over 500 monitoring stations were selected that most nearly represented sites with adequate data and free from urban influence. Stations and observations were screened to eliminate those with few or unrepresentative readings. The averaging technique produced monthly means of the maximum 7-h/d ozone levels for each station. A daily curve was calculated based on the hourly observations between 8:00 a.m. and 8:00 p.m., with the maximum 7-h/d value estimated from the curve. Monthly means were estimated for stations having at least 10 daily values.

The monitoring stations are irregularly distributed with large areas of the country lacking monitoring data for rural areas. The available data were used to estimate county values using a Kriging (J. A. Reagan, personal communication) analysis. The method used an inverse distance-squared weighting factor so that estimates from the nearest monitoring stations were given the most weight in determining county values. The temporal variance (variability within a month) and the spatial variance (variability of stations used to estimate the county value) were calculated. The Los Angeles basin counties were excluded from the analysis due to high variability of monitoring data occurring in the large county units. Seasonal averages were calculated as the mean of the growing season months appropriate for each crop (wheat, April–May; corn, peanuts, June–August; and soybeans, June–September). The patterns for each growing season are mapped in Figs. 1 to 3, respectively.

ANALYSIS

The crop loss functions were obtained by first deriving relations between crop yield and seasonal 7-h/d ozone concentrations for each of the 9 separate data sets. Several different regression models were fitted to each data set.* Although the regressions are based on a limited number of cultivars for each crop grown under ideal test conditions, they provide a realistic estimate of crop yield responses to ozone. Complex models only improved the fit of one of the four winter wheat data sets and peanuts, it was decided to develop the loss functions from simple linear models.

These ozone dose-yield response relations for each of the 9 data sets were converted to percent-yield reduction models by scaling to a control yield predicted at 25 ppb ozone. 25 ppb ozone approximates the 7-h/d seasonal concentration in the charcoal-filtered air. (See Table 2. Yields predicted at doses of 60 and 100 ppb are also shown.) These individual percent-yield reduction models were grouped by crop type (across cultivars). The number of chamber yields comprising the response to a particular dose ranged from two for the two years of Davis soybean data to five for corn. Dose in all models is reported as the 7-h/d seasonal mean in parts per billion (ppb) and yields

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* The goal was the simplest set of models that adequately fit the data to facilitate the eventual combining of the individual models across cultivars and across species. More complex models were pursued only if they significantly improved the fit to the data relative to the simplest model. Even then, consideration was given to the relative importance of the particular crop or cultivar and the magnitude of the change in predicted yields associated with the more complex model.

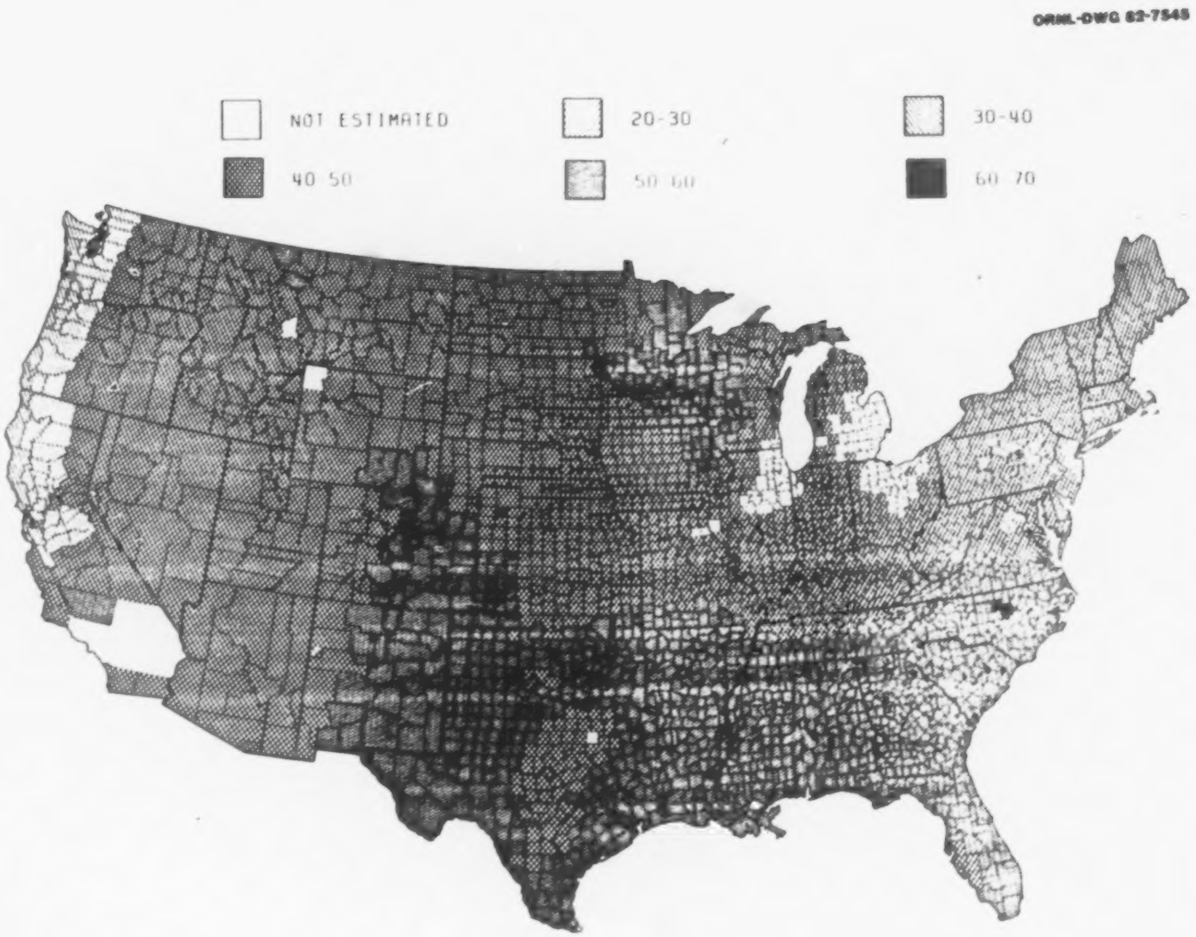
ORNL-DWG 82-7545
☐ NOT ESTIMATED
☐ 20-30
▨ 30-40
▨ 40-50
▨ 50-60
■ 60-70

Fig. 1. April-May average county-level ozone concentrations (maximum 7-h/d concentrations - ppb) based on 1978 monitoring data, estimation procedure by J. A. Reagan, EPA.

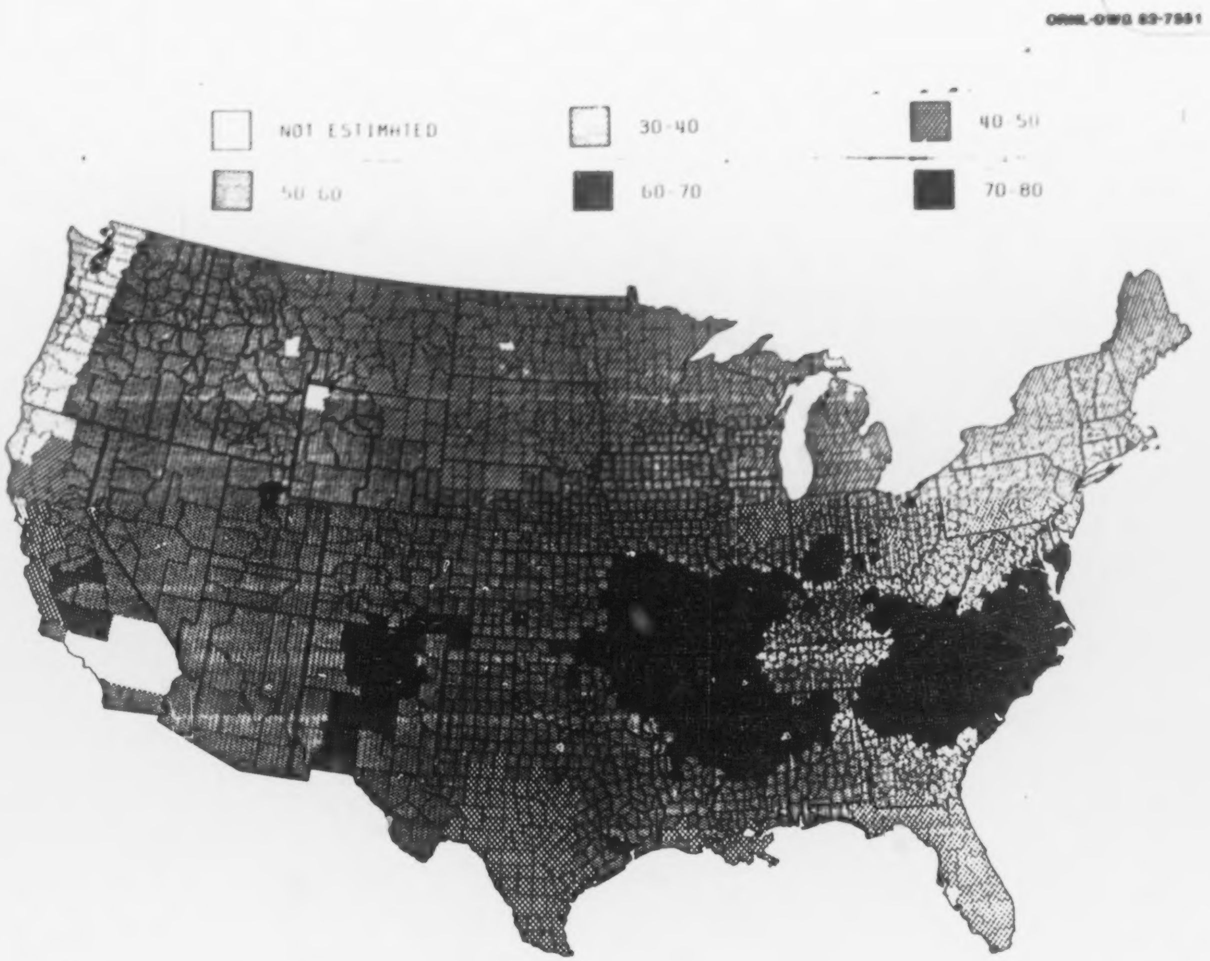
ORNL-DWG 83-7581
NOT ESTIMATED
30-40
40-50
50-60
60-70
70-80

Fig. 2. June-September average county-level ozone concentrations (maximum 7-h/d concentrations - ppb) based on 1978 monitoring data, estimation procedure by J. A. Reagan, EPA.

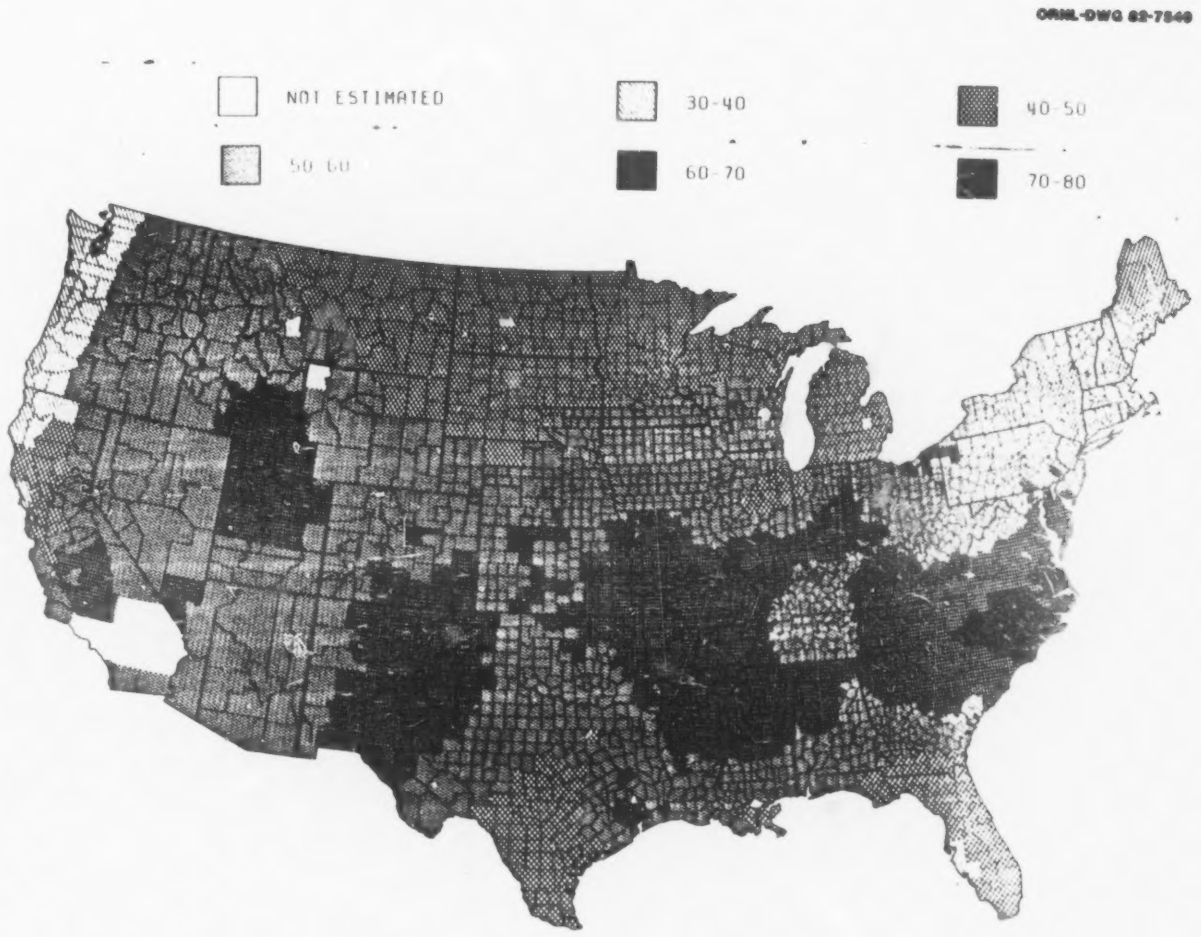
ORNL-DWG 82-7549
NOT ESTIMATED
30-40
40-50
50-60
60-70
70-80

Fig. 3. June-August average county-level ozone concentrations (maximum 7-h/d concentrations - ppb) based on 1978 monitoring data, estimation procedure by J. A. Reagan, EPA.

Table 2
SIMPLE REGRESSION MODELS OF YIELD (y) IN G/PLANT AS A FUNCTION OF 7-H/D OZONE DOSE IN PPB

Crop (cultivar)	Regression model of form $y = b_0 + b_1x$. Values in parentheses are ± standard error for that parameter[a]	R^2	Predicted yield in g/plant (± SE)[b]		
			(25 ppb)	(60 ppb)	(100 ppb)
Corn (Coker 16)	y = 247.829 - 0.260x (±6.342) (±0.068)	0.65	241.3 (±5.2)	232.2 (±3.6)	221.8 (±3.4)
Soybean (Corsoy)	y = 23.490 - 0.126x (±0.699) (±0.010)	0.96	20.3 (±0.5)	15.9 (±0.3)	10.9 (±0.5)
Soybean (Davis 1977)	y = 96.574 - 0.385x (±7.538) (±0.078)	0.90	87.0 (±5.9)	73.5 (±4.0)	58.1 (±3.6)
Soybean (Davis 1978)	y = 95.336 - 0.309x (±9.254) (±0.109)	0.99	87.6 (±6.9)	76.8 (±4.4)	64.4 (±4.7)
Peanut (NC-6)	y = 173.195 - 1.046x (±3.565) (±0.042)	0.99	147.1 (±2.6)	110.4 (±1.6)	68.6 (±1.8)
Wheat (Blueboy II)	y = 6.609 - 0.018x (±0.258) (±0.003)	0.92	6.1 (±0.2)	5.5 (±0.1)	4.8 (±0.1)
Wheat (Coker 47-27)	y = 5.795 - 0.021x (±0.199) (±.002)	0.98	5.3 (±0.2)	4.5 (±0.1)	3.7 (±0.1)
Wheat (Holly)	y = 5.668 - 0.015x (±0.245) (±0.003)	0.82	5.3 (±0.2)	4.7 (±0.1)	4.1 (±0.1)
Wheat (Oasis)	y = 4.880 - 0.011x (±0.206) (±.002)	0.88	4.6 (±0.2)	4.2 (±0.1)	3.7 (±0.1)

[a] Standard error term corrected from regression analysis value based on regression error term to that based on AOV mean square error:

$$SE(b_i) = [(\text{AOV MSE}/\text{Reg.MSE})]^{1/2}$$

[b] Standard error computed as follows:

$$SE(y_{x=x_i}) = \{S^2 [\frac{1}{n} + \frac{(x_i - \bar{x})^2}{\Sigma x^2}]\}^{1/2}$$

where S^2 is the mean square error from the AOV, n the number of observations, x_i the selected dose (25, 60, or 100 ppb), $\bar{x}$ the mean of the doses, and Σx^2 the corrected dose sums of squares.

are grams/plant.

Regional Impacts

The impacts of ozone are already reflected in the 1978 Census of Agriculture yield data. That is, the actual yield represents a potential yield minus a loss due to the ambient ozone and other environmental factors. Therefore, the crop loss functions were used to calculate the potential yield assuming reduction in ozone levels. These values were calculated by dividing current yields by a reduced-yield factor, where the reduced-yield factor was calculated using the crop loss functions and current ozone levels. Crop gains were than calculated as the difference between current yields and the estimated yield assuming a reduction in ozone levels to background concentrations (25 ppb).

RESULTS

The maps of crop gains show the general pattern of impact for each crop (Figs. 4-7) in terms of changes in productivity (i.e., kg/ha). Tables 3-6 summarize the changes by state. The tables give statistics in terms of yield and dollar value (1978 average market price), not taking into consideration any potential price effect. An economic analysis of the data would have to include an assessment of the effect of yield and dollar value (1978 average market price), not taking into consideration any potential price effect. An economic analysis of the data would have to include an assessment of the effect of yield gains on crop market structure. Table 7 summarizes the dollar values of the crop gains for all four crops considered.

The assessment estimates that approximately three billion dollars of productivity could be gained if current maximum 7-h/d ozone levels were

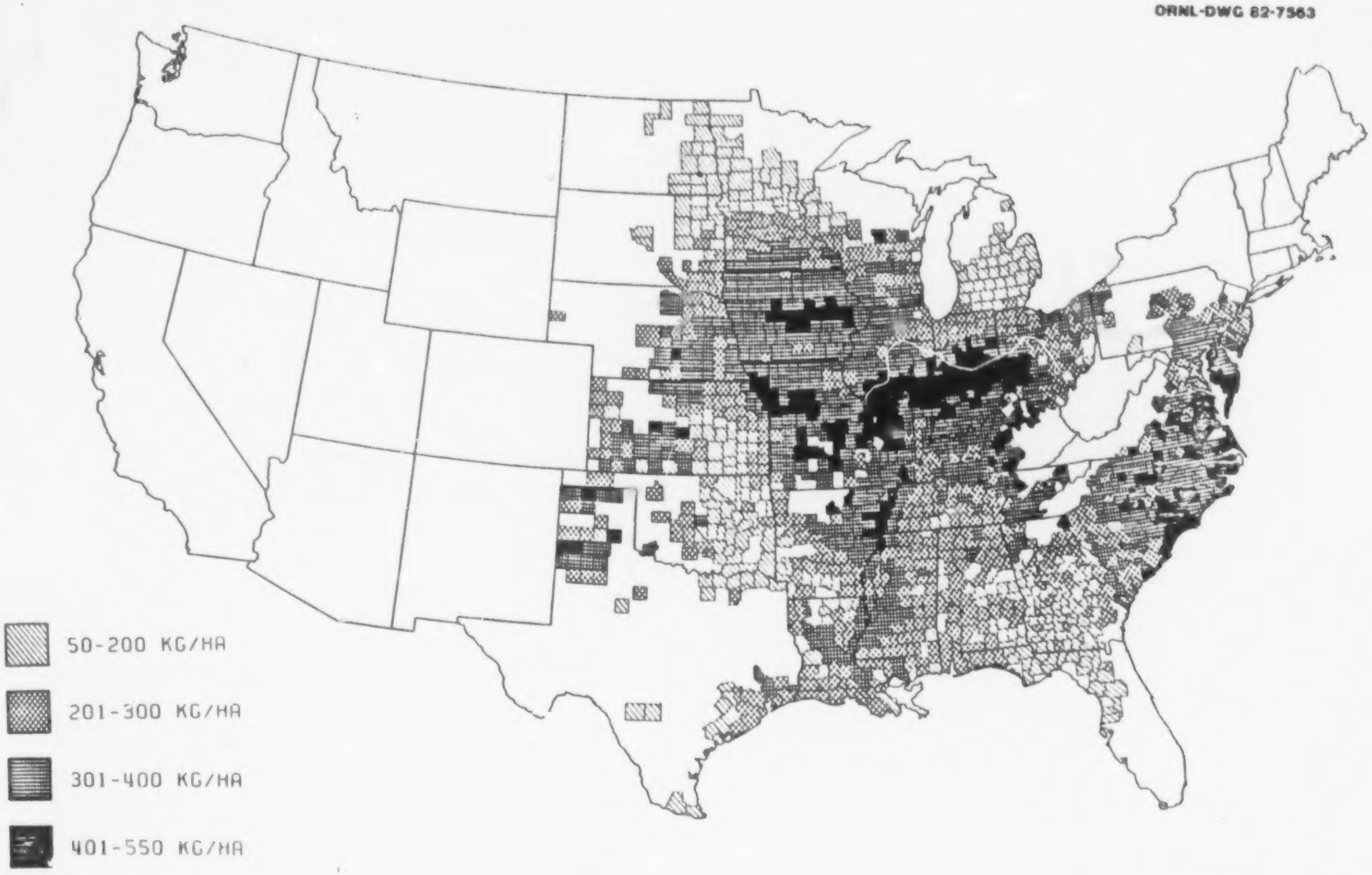
ORNL-DWG 82-7563
50-200 KG/HA
201-300 KG/HA
301-400 KG/HA
401-550 KG/HA

Fig. 4. Soybean yield gains (kg/ha) with ozone reductions based on 1978 agriculture census and 1978 EPA ozone data.

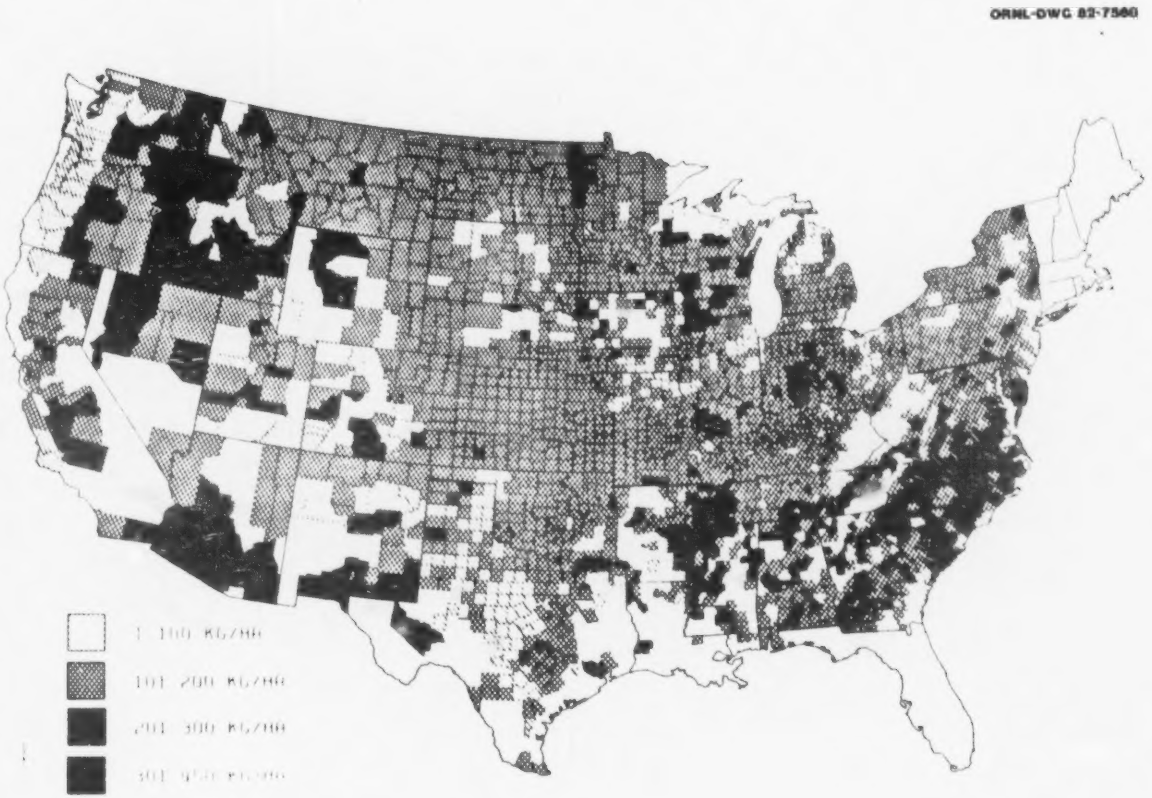
ORNL-DWG 82-7580
1 100 KG/HA
101 200 KG/HA
201 300 KG/HA
301 [illegible] KG/HA

Fig. 5. Wheat yield gains (kg/ha) with ozone reductions based on 1978 agriculture census and 1978 EPA ozone data.

ORNL-DWG 82-7561

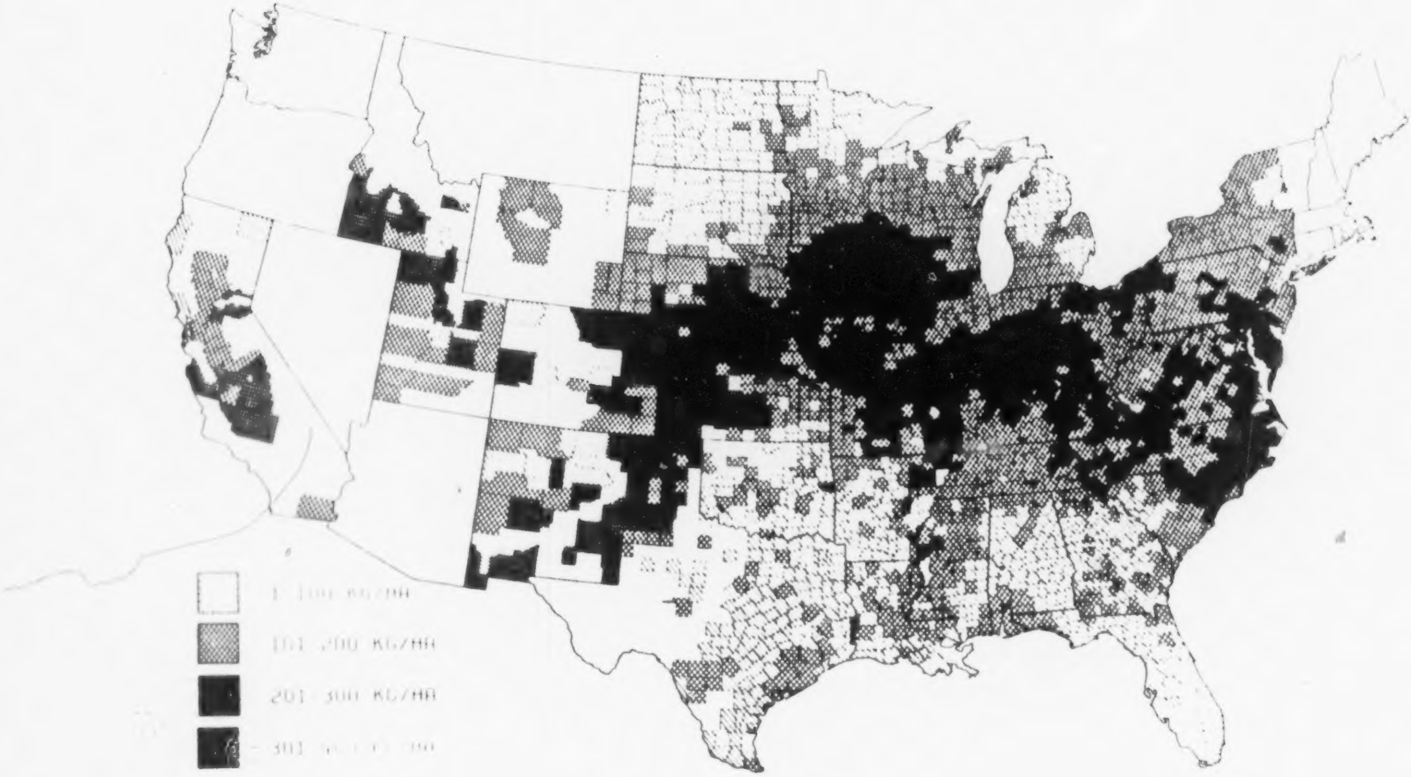
1-100 KG/HA
101-200 KG/HA
201-300 KG/HA
301

Fig. 6. Corn yield gains (kg/ha) with ozone reductions based on 1978 agriculture census and 1978 EPA ozone data.

ORNL-DWG 82-7565

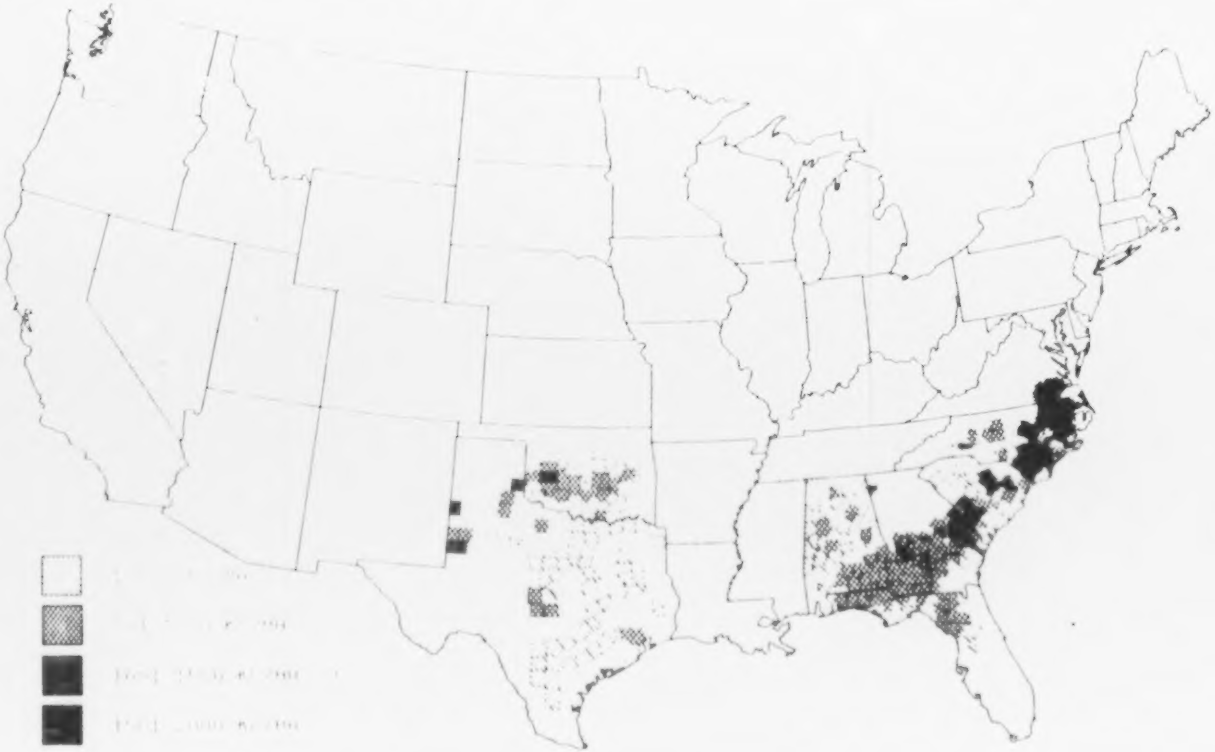

Fig. 7. Peanut yield gains (kg/ha) with ozone reductions based on 1978 agriculture census and 1978 EPA ozone data.

Table 3

WHEAT PRODUCTION AND POTENTIAL GAINS WITH OZONE REDUCTIONS BASED ON 1978 CROP DATA, 1978 PRICE OF $2.98/BU, AND 1978 EPA OZONE DATA

State	Percent gain	1978 yield bu x 10^6	Gain bu x 10^6	New yield bu x 10^6	1978 value $ x 10^6	Gain $ x 10^6	New value $ x 10^6
Alabama	10.7	1	0	2	4	0	5
Arizona	7.7	8	1	8	22	2	24
Arkansas	9.2	9	1	10	26	2	98
California	5.6	31	2	33	93	5	98
Colorado	9.0	55	5	60	164	15	178
Connecticut	0.0	0	0	0	0	0	0
Delaware	7.8	1	0	1	2	0	2
Florida	0.0	0	0	0	0	0	0
Georgia	10.6	3	0	3	9	1	10
Idaho	7.8	69	5	75	208	16	222
Illinois	7.1	31	2	34	93	7	100
Indiana	7.1	22	2	24	66	5	71
Iowa	7.1	1	0	1	3	0	3
Kansas	7.8	269	21	290	800	63	863
Kentucky	7.0	6	0	6	18	1	19
Louisiana	10.3	0	0	0	1	0	1
Maine	0.0	0	0	0	0	0	0
Maryland	6.6	3	0	3	8	1	8
Massachusetts	0.0	0	0	0	0	0	0
Michigan	4.6	15	1	15	44	2	46
Minnesota	8.8	81	7	88	241	20	261
Mississippi	9.8	2	0	2	5	1	6
Missouri	7.2	27	2	29	81	6	86
Montana	8.1	134	11	145	399	32	432
Nebraska	7.7	73	6	78	217	17	234
Nevada	7.4	1	0	1	3	0	3
New Hampshire	0.0	0	0	0	0	0	0
New Jersey	4.6	1	0	1	2	0	2
New Mexico	11.3	5	1	6	16	2	13
New York	6.7	2	0	2	7	0	7
North Carolina	11.1	4	0	5	13	1	14
North Dakota	7.8	268	21	289	800	62	862
Ohio	5.2	37	2	38	109	6	115
Oklahoma	8.6	125	11	135	371	32	403
Oregon	6.5	46	3	49	136	9	145
Pennsylvania	8.3	6	1	7	19	2	20
Rhode Island	0.0	0	0	0	0	0	0
South Carolina	10.7	2	0	2	5	1	6
South Dakota	7.8	64	5	69	191	15	206
Tennessee	8.7	5	0	6	15	1	16
Texas	9.4	55	5	60	164	15	179
Utah	7.9	7	1	7	20	2	21
Vermont	0.0	0	0	0	0	0	0
Virginia	8.1	4	0	4	12	1	13
Washington	7.7	124	10	133	368	28	397
West Virginia	6.5	0	0	0	1	0	1
Wisconsin	7.1	2	0	0	1	0	1
Wyoming	8.4	6	1	7	19	2	21
United States	7.8	1,604	126	1,729	4,779	375	5,153

Table 4

SOYBEAN PRODUCTION AND POTENTIAL GAINS WITH OZONE REDUCTIONS BASED ON 1978 CROP DATA, 1978 PRICE OF $6.66/BU, AND 1978 EPA OZONE DATA

State	Percent gain	1978 yield bu x 10^6	Gain bu x 10^6	New yield bu x 10^6	1978 value \$ x 10^6	Gain \$ x 10^6	New value \$ x 10^6
Alabama	18.4	31	6	37	208	38	246
Arizona	0.0	0	0	0	0	0	0
Arkansas	22.9	106	24	130	703	161	864
California	0.0	0	0	0	0	0	0
Colorado	0.0	0	0	0	0	0	0
Connecticut	0.0	0	0	0	0	0	0
Delaware	19.6	7	1	9	50	10	59
Florida	13.5	8	1	9	52	7	59
Georgia	17.7	27	5	32	180	32	212
Idaho	0.0	0	0	0	0	0	0
Illinois	16.9	301	51	352	2,005	339	2,345
Indiana	17.9	135	24	159	898	161	1,059
Iowa	15.2	266	40	306	1,771	269	2,039
Kansas	20.3	26	5	32	176	36	212
Kentucky	17.2	36	6	42	238	41	279
Louisiana	16.1	75	12	87	501	81	581
Maine	0.0	0	0	0	0	0	0
Maryland	19.9	11	2	14	77	15	92
Massachusetts	0.0	0	0	0	0	0	0
Michigan	9.8	21	2	23	141	14	155
Minnesota	12.6	123	15	138	816	103	919
Mississippi	21.7	74	16	90	491	107	598
Missouri	20.3	141	29	170	938	191	1,129
Montana	0.0	0	0	0	0	0	0
Nebraska	15.3	38	6	43	251	38	289
Nevada	0.0	0	0	0	0	0	0
New Hampshire	0.0	0	0	0	0	0	0
New Jersey	13.8	5	1	6	35	5	40
New Mexico	0.0	0	0	0	0	0	0
New York	0.0	0	0	0	0	0	0
North Carolina	26.0	35	9	44	234	61	294
North Dakota	9.8	4	0	5	28	3	30
Ohio	15.4	121	19	140	806	124	930
Oklahoma	20.1	5	1	6	33	7	40
Oregon	0.0	0	0	0	0	0	0
Pennsylvania	14.6	2	0	3	15	2	18
Rhode Island	0.0	0	0	0	0	0	0
South Carolina	23.5	29	7	36	194	46	240
South Dakota	12.9	11	1	13	76	10	86
Tennessee	19.0	49	9	58	327	62	389
Texas	17.6	16	3	18	104	18	122
Utah	0.0	0	0	0	0	0	0
Vermont	0.0	0	0	0	0	0	0
Virginia	22.9	12	3	15	79	18	97
Washington	0.0	0	0	0	0	0	0
West Virginia	0.0	0	0	0	0	0	0
Wisconsin	13.6	7	1	7	44	6	50
Wyoming	0.0	0	0	0	0	0	0
United States	17.5	1,722	301	2,023	11,471	2,003	13,474

Table 5

CORN PRODUCTION AND POTENTIAL GAINS WITH OZONE REDUCTIONS BASED ON 1978 CROP DATA, 1978 PRICE OF \$2.25/BU, AND 1978 EPA OZONE DATA

State	Percent gain	1978 yield bu x 10^6	Gain bu x 10^6	New yield bu x 10^6	1978 value $\$$ x 10^6	Gain $\$$ x 10^6	New value $\$$ x 10^6
Alabama	3.5	23	1	24	51	2	53
Arizona	0.0	0	0	0	0	0	0
Arkansas	4.7	1	0	1	3	0	3
California	2.8	33	1	34	74	2	76
Colorado	3.7	78	3	80	174	6	181
Connecticut	0.0	0	0	0	0	0	0
Delaware	4.1	15	1	16	34	1	35
Florida	2.5	15	0	15	34	1	35
Georgia	3.1	77	2	79	173	5	178
Idaho	3.9	5	0	5	11	0	11
Illinois	3.3	1,182	39	1,221	2,660	87	2,747
Indiana	3.8	627	24	650	1,410	53	1,463
Iowa	3.4	1,390	47	1,437	3,127	106	3,233
Kansas	3.9	152	6	158	342	13	356
Kentucky	3.7	110	4	114	248	9	257
Louisiana	3.5	2	0	2	4	0	4
Maine	0.0	0	0	0	0	0	0
Maryland	4.0	56	2	58	125	5	130
Massachusetts	0.0	0	0	0	0	0	0
Michigan	2.3	185	4	189	416	10	425
Minnesota	2.8	570	16	586	1,282	36	1,318
Mississippi	4.1	5	0	5	11	0	11
Missouri	4.2	183	8	191	412	17	430
Montana	0.0	0	0	0	0	0	0
Nebraska	3.5	705	25	730	1,587	56	1,643
Nevada	0.0	0	0	0	0	0	0
New Hampshire	0.0	0	0	0	0	0	0
New Jersey	3.0	8	0	8	18	1	18
New Mexico	4.2	6	0	7	14	1	15
New York	3.0	51	2	52	114	3	117
North Carolina	5.3	114	6	120	256	14	269
North Dakota	2.2	20	0	21	45	1	46
Ohio	3.4	351	12	363	790	27	817
Oklahoma	4.0	4	0	4	8	0	9
Oregon	0.0	0	0	0	0	0	0
Pennsylvania	3.4	114	4	117	256	9	264
Rhode Island	0.0	0	0	0	0	0	0
South Carolina	4.7	29	1	31	66	3	69
South Dakota	2.7	169	5	173	379	10	390
Tennessee	3.8	42	2	44	95	4	98
Texas	3.8	115	4	119	258	10	268
Utah	4.2	1	0	1	3	0	3
Vermont	0.0	0	0	0	0	0	0
Virginia	4.4	47	2	49	105	5	110
Washington	0.0	0	0	0	0	0	0
West Virginia	3.2	6	0	6	13	0	13
Wisconsin	3.0	289	9	297	649	19	668
Wyoming	3.1	3	0	3	6	0	6
United States	3.4	6,778	230	7,008	15,251	517	15,769

TABLE 6.

PEANUT PRODUCTION AND POTENTIAL GAINS WITH OZONE REDUCTIONS BASED ON 1978 CROP DATA, 1978 PRICE OF $0.21/LB, AND 1978 EPA OZONE DATA

State	Percent gain	1978 Yield bu x 10^6	Gain bu x 10^6	New yield bu x 10^6	1978 value $ x 10^6	Gain $ x 10^6	New value $ x 10^6
Alabama	27.1	500	137	642	106	29	135
Arizona	0.0	0	0	0	0	0	0
Arkansas	0.0	0	0	0	0	0	0
California	0.0	0	0	0	0	0	0
Colorado	0.0	0	0	0	0	0	0
Connecticut	0.0	0	0	0	0	0	0
Delaware	0.0	0	0	0	0	0	0
Florida	20.6	152	31	184	32	7	39
Georgia	24.7	1,546	382	1,928	325	80	405
Idaho	0.0	0	0	0	0	0	0
Illinois	0.0	0	0	0	0	0	0
Indiana	0.0	0	0	0	0	0	0
Iowa	0.0	0	0	0	0	0	0
Kansas	0.0	0	0	0	0	0	0
Kentucky	0.0	0	0	0	0	0	0
Louisiana	0.0	0	0	0	0	0	0
Maine	0.0	0	0	0	0	0	0
Maryland	0.0	0	0	0	0	0	0
Massachusetts	0.0	0	0	0	0	0	0
Michigan	0.0	0	0	0	0	0	0
Minnesota	0.0	0	0	0	0	0	0
Mississippi	0.0	0	0	0	0	0	0
Missouri	0.0	0	0	0	0	0	0
Montana	0.0	0	0	0	0	0	0
Nebraska	0.0	0	0	0	0	0	0
Nevada	0.0	0	0	0	0	0	0
New Hampshire	0.0	0	0	0	0	0	0
New Jersey	0.0	0	0	0	0	0	0
New Mexico	0.0	0	0	0	0	0	0
New York	0.0	0	0	0	0	0	0
North Carolina	48.3	445	215	659	93	45	138
North Dakota	0.0	0	0	0	0	0	0
Ohio	0.0	0	0	0	0	0	0
Oklahoma	30.2	177	54	231	37	11	48
Oregon	0.0	0	0	0	0	0	0
Pennsylvania	0.0	0	0	0	0	0	0
Rhode Island	0.0	0	0	0	0	0	0
South Carolina	41.8	31	13	44	7	3	9
South Dakota	0.0	0	0	0	0	0	0
Tennessee	0.0	0	0	0	0	0	0
Texas	21.5	394	85	478	83	18	100
Utah	0.0	0	0	0	0	0	0
Vermont	0.0	0	0	0	0	0	0
Virginia	47.5	255	121	376	54	25	79
Washington	0.0	0	0	0	0	0	0
West Virginia	0.0	0	0	0	0	0	0
Wisconsin	0.0	0	0	0	0	0	0
Wyoming	0.0	0	0	0	0	0	0
United States	29.6	3,505	1,037	4,542	736	218	954

TABLE 7.

POTENTIAL GAINS (MILLIONS OF DOLLARS) WITH OZONE REDUCTIONS BASED ON 1978 CROP PRODUCTION, PRICES, AND OZONE DATA

State	Wheat	Corn	Soybean	Peanuts	All crops
Alabama	0	2	38	29	69
Arizona	2	0	0	0	2
Arkansas	2	0	161	0	163
California	5	2	0	0	7
Colorado	15	6	0	0	21
Connecticut	0	0	0	0	0
Delaware	0	1	10	0	11
Florida	0	1	7	7	14
Georgia	1	5	32	80	118
Idaho	16	0	0	0	17
Illinois	7	87	339	0	433
Indiana	5	53	161	0	218
Iowa	0	106	269	0	374
Kansas	63	13	36	0	112
Kentucky	1	9	41	0	51
Louisiana	0	0	81	0	81
Maine	0	0	0	0	0
Maryland	1	5	15	0	21
Massachusetts	0	0	0	0	0
Michigan	2	10	14	0	25
Minnesota	20	36	103	0	159
Mississippi	1	0	107	0	107
Missouri	6	17	191	0	214
Montana	32	0	0	0	32
Nebraska	17	56	38	0	111
Nevada	0	0	0	0	0
New Hampshire	0	0	0	0	0
New Jersey	0	1	5	0	5
New Mexico	2	1	0	0	2
New York	0	3	0	0	4
North Carolina	1	14	61	45	121
North Dakota	62	1	3	0	66
Ohio	6	27	124	0	157
Oklahoma	32	0	7	11	50
Oregon	9	0	0	0	9
Pennsylvania	2	9	2	0	12
Rhode Island	0	0	0	0	0
South Carolina	1	3	46	3	52
South Dakota	15	10	10	0	35
Tennessee	1	4	62	0	67
Texas	15	10	18	18	61
Utah	2	0	0	0	2
Vermont	0	0	0	0	0
Virginia	1	5	18	25	49
Washington	28	0	0	0	28
West Virginia	0	0	0	0	0
Wisconsin	0	19	0	0	26
Wyoming	2	0	0	0	2
United States	375	517	2,003	218	3,112

reduced below 25 ppb. Dollar values are based on 1978 crop prices, without accounting for price effects, to provide an overall estimate of the impact. Of the estimated dollar impact, soybean represents 64%, corn 17%, wheat 12%, and peanuts 7%. The corn-belt states of Illinois, Iowa, and Indiana experienced the greatest impacts. Missouri, Arkansas, Minnesota, and Ohio also had high impacts.

DISCUSSION AND CONCLUSIONS

The dose-response functions predict quantitative changes in economic yield whereas previous studies have been limited to observing foliar changes. Both ozone and crop data refer to 1978. Thus the assessment provides maps and statistical summaries of ozone impacts on crops using county-level resolution of ozone and crop data with realistic dose-response relationships.*

Future assessments may very well develop more complex statistical models that will more adequately represent biological response as well as more effectively deal with the issue of uncertainty analysis associated with the aggregation of large data bases used in assessments of this type.

Assumptions and caveats as discussed must be considered when interpreting the level of uncertainty associated with this or other similar assessments. However, important considerations can be drawn from the assessment. First, it is readily apparent from the ozone maps that a significant portion of the agricultural land area of the United States is experiencing levels of oxidant during the growing season capable of reducing crop yields. Secondly, potential impacts as determined by this method further support earlier, independent estimates by Heagle and Heck (1980) which suggested annual losses of approximately two billion dollars. The method described demonstrates the utility of county-level data for integrated regional-scale impact assessment studies and the need for experimental data representative of regional agricultural practices. Finally, statistical techniques need to be developed which will estimate the uncertainty of the estimated impacts by combining the sampling errors associated with each of the component data sets.

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* It is important to note that for the purposes of this analysis linear models were used to fit the dose response for all crops. For the corn data used, the quadratic term in the dose-response model was nonsignificant. Under those conditions, use of the quadratic model would result in an unrealistic increase in yield from 25-40 pbb. As a result, both the linear and quadratic models, by virtue of their simplicity, are suboptimal representations of the biological response of corn.

The NCLAN studies are expanding the number of crops, cultivars, and environmental factors being measured. These additional data will allow the dose-response functions to be refined for future asessments. Environmental data, such as relative humidity and moisture status (evaporation vs. precipitation) are available at the county level that could be incorporated into the analysis. The network of monitoring stations measuring ozone, especially in rural areas, is expanding. Thus the ozone data base is becoming more accurate. Also, with these estimated changes in yields, an economic analysis could be undertaken to consider price effects to refine the dollar value associated with reducing ozone levels.

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2. EFFECTS OF ACID RAIN ON AGRICULTURAL CROPS

INTRODUCTION

Acidic precipitation is frequently cited as a potential threat to agricultural productivity in the United States because patterns of deposition of elevated levels of hydrogen ions as "acid rain" coincide with major regions of agricultural productivity. The map (Figure 2.1) illustrates those agricultural regions (counties with >30% agricultural land area) that experience elevated hydrogen ion deposition as well as ozone.

Mechanisms for an effect are currently thought to involve direct effect on above-ground portions of the plant rather than some soil-mediated effect. Potential interactions with other physical and biotic causes of stress are hypothesized, but little definitive evidence exists on which to base conclusions.

It is important to note that (1) significant research efforts on acidic precipitation effects are less than 5 years old -- approximately the same stage of development as research on ozone effects on vegetation was in the late 1960s, (2) acidic precipitation effects involve a dose response not only to hydrogen ion but also to sulfate and nitrate, and (3) effects may be manifest not only on aboveground portions of the plant but also through soil-mediated effects on nutrient availability, microbial activity, or perhaps even metal toxicity.

SOURCES OF AVAILABLE DATA

Mechanisms of Acid Rain Effect on Vegetation

Rain, fog, dew, and other forms of wet deposition that are not

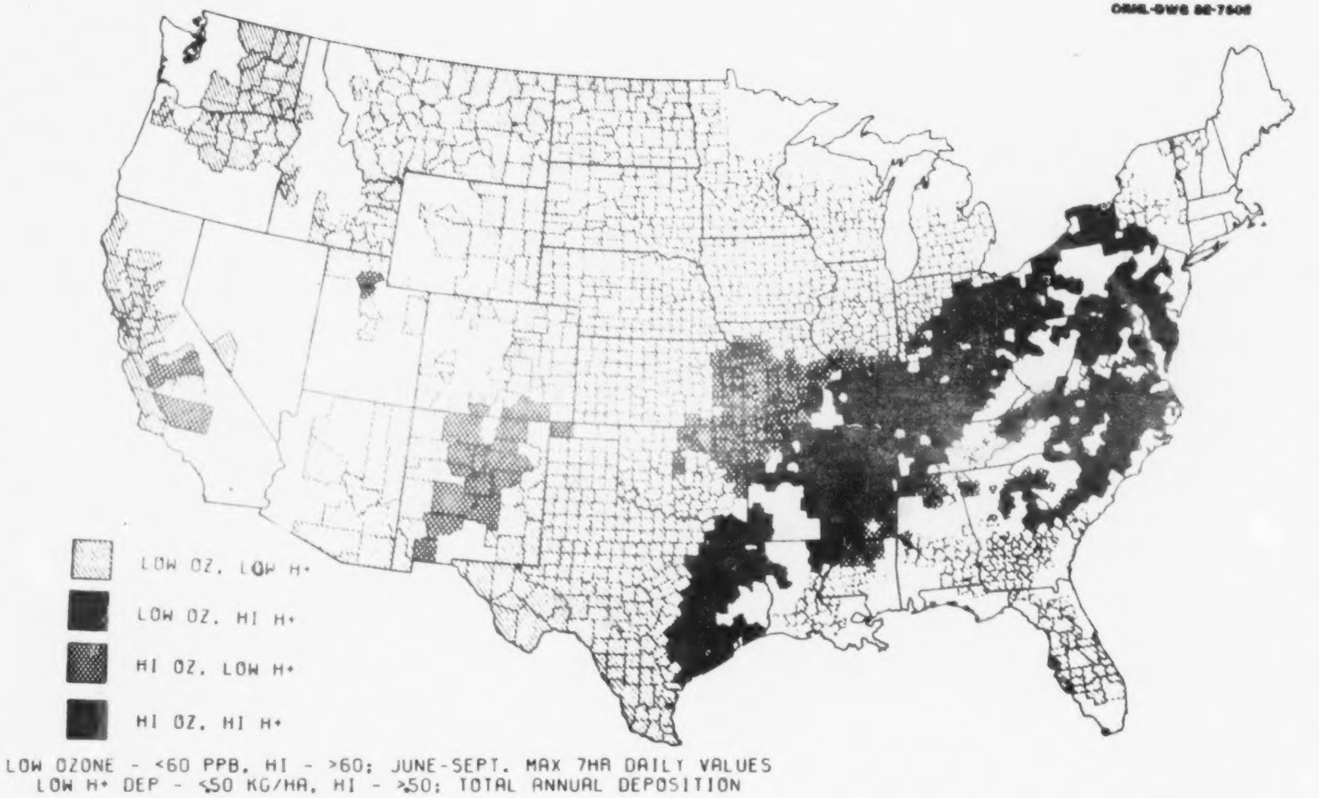
ORNL-DWG 82-7608
LOW OZ. LOW H+
LOW OZ. HI H+
HI OZ. LOW H+
HI OZ. HI H+
LOW OZONE - <60 PPB, HI - >60; JUNE-SEPT. MAX 7HR DAILY VALUES
LOW H+ DEP - ≤50 KG/HA, HI - ≥50; TOTAL ANNUAL DEPOSITION

Fig. 2.1 Ozone and acid rain levels for agricultural areas (>30%) 1978 EPA ozone data.

contaminated with pollutants play an important role for vegetation and soils as sources of nutrient inputs.

There are no documented cases of direct, visible injury to vegetation in the field situation as a result of exposure to "acid rain". Rather, information about effects is the result of a wide variety of approaches involving, in most instances, the use of some form of rain "simulation." It is clear from the evidence available that wet deposition of pollutants may result in a range of direct effects on vegetation, depending on (1) precedent conditions (e.g., soil nutrient status, plant nutrient requirements, plant sensitivity, and growth stage) and (2) the total loading or deposition of critical ions (H^+, NO_3, SO_4).

Increased inputs of H^+, SO_4^-, and NO_3^- have both positive and negative effects on soil-plant systems. Concentrations of hydrogen ion, equivalent to those measured in more acidic rain events ($\leq$pH3), are observed experimentally to cause tissue injury in the form of necrotic lesions to a wide variety of plant species under greenhouse and laboratory conditions. The threshold for the occurrence of this visible injury is reported as being between pH 3.0 and 3.6. The various types of direct effects which have been reported are shown in Table 2.1. Such results must be interpreted with caution, however, because the growth and morphology of leaves in a greenhouse are often atypical of field conditions. No evidence at the present time suggests any positive or beneficial effect of hydrogen ion inputs per se.

Recent research identified the extremely important role of sulfur and nitrogen inputs in precipitation to the ultimate response of vegetation to acid deposition. Evidence from studies of field-grown soybeans (Irving 1979) and forest tree species (Wood and Bormann (1974) and Abrahamson and Dollard

Table 2.1 Types of direct, visible injury reported in response to acidic wet deposition

Injury type	Species	pH range	Reference	Remarks
Pitting, curl shortening, death	Yellow birch	2.3-4.7	Wood and Bormann (1974)	
1-mm necrotic lesions, premature abscission	Kidney bean, soybean, loblolly pine, E. white pine, willow oak	3.2	Shriner et al. (1974)	
Cuticular erosion	Willow oak	3.2	Shriner (1978a), Lang et al. (1978)	
Chlorosis	Sunflower, bean	2.3-5.7	Evans et al. (1977)	
(A) small, shallow circular depressions: slight chlorosis	Sunflower, bean	2.7	Evans et al. (1977)	More frequent near veins. (A) - (D) represent sequential stages of lesion development, through time, up to 72 h (one 6-min rain event daily for 3 d)
(B) larger lesions chlorosis always present palisade collapse	Sunflower, bean	2.7	Evans et al. (1977)	
(C) 1-mm necrotic lesions general distortion	Sunflower, bean	2.7	Evans et al. (1977)	
(D) 2-mm bifacial due to coalescence of smaller lesions, total tissue collapse.	Sunflower, bean	2.7	Evans et al. (1977)	
Wrinkled leaves, excessive adventitious budding, premature absission	Bean	1.5-3.0	Ferenbaugh (1976)	
Incipient bronzed spot	Bean	2.0-3.0	Hundawi et al. (1980)	After first few hours
Bifacial necrotic	Bean	2.0-3.0	Hindawi et al. (1980)	After 24 h (reported pooling of drops = more injury)
Necrotic lesions, premature abscission	E. White pine, scotch pine, spinach, sunflower, bean	2.6-3.4	Jacobson and van Leuken (1977)	Injury associated with droplet location within 24-48 h
Marginal and tip necrosis	Bean, poplar, soybean, ash, birch, corn, wheat	Submicron H_2SO_4 aerosol	Lang et al. (1978)	
Galls, hypertrophy, hyperplasia	Hybrid poplar	2.7-3.4	Evans et al. (1978)	
Dead leaf cells	Soybean	3.1	Irving (1979)	
Necrotic lesions	Citrus	0.5-2.0	Heagle et al. (1978)	

(1979) exposed to simulated acid rains indicate an apparent positive growth response from sulfur and nitrogen inputs in simulated acid rain. Turner and Lambers (1980) presented evidence of a positive growth response in Monterey pine to sulfur deposition in ambient precipitation in Australia, and work by Jacobsen (1980b) suggests that sulfur may have been a limiting factor in the nutrition of experimental lettuce plots.

Soil properties such as cation exchange capacity, base saturation, soil depth, and sulfate adsorption (Reuss 1978, Norton 1980, McFee 1980) are important factors in the long-term impacts of wet deposition of pollutants in soil-plant systems. However, since croplands are heavily managed, such micronutrient deficiencies due to acid deposition are unlikely.

Dose Response

Several investigators reported results of dose-response experiments with a number of different plant species. Thresholds for direct, visible injury to foliage in greenhouse studies with acid rains are typically about pH 3.1 (Jacobson et al. 1980, Shriner et al. 1974). However, in field trials where the same investigators used the same treatment solutions under both greenhouse and field situations there are significant shifts in species sensitivity.

"Wettability" appears to be an important factor in the response of plants to acid deposition. This can be seen in the work of Evans et al. (1977), Jacobson and Van Leuken (1977), and Shriner (1978a) who variously report a threshold of between pH 3.1 and 3.5 for development of foliar lesions on beans. The cultivars used in these studies are all relatively nonwaxy and therefore fairly easily wettable. By comparison, studies with the very waxy leaves of citrus (Heagle et al. 1978) reported a threshold for visible symptoms to near pH 2.0. Waxy leaves apparently minimize the contact time for acid solutions, thus accounting for the >400X increase in H^+ concentration required to induce visible injury. Table 2.2 summarizes the thresholds, including range, species sensitivity, concentration, and time, for visible injury associated with experimental studies of wet deposition of acidic substances.

To summarize our knowledge of dose response: (1) visible injury thresholds for acid precipitation lie between pH 2.0 and 3.6, depending on species, and may vary from pH 3.0 to 3.6 within the same species (e.g., bean); (2) species-to-species variability in sensitivity may be as great as two orders of magnitude in hydrogen ion concentration; (3) the threshold for growth effects in the absence of visible injury is reported between pH 3.5 and 4.0, but sulfur and nitrogen inputs in the precipitation may result in a net positive growth impact, depending on soil nutrient status, buffer capacity, other growth conditions, and plant nutrient requirements, and (4) total dose of hydrogen ions appears to be most clearly related to visible injury.

RELATIONSHIP OF ACID RAIN TO OVERALL CROP DAMAGE FROM POLLUTANTS

Because of the widespread occurrence of wet deposition of acidic substances in the eastern United States and in northern Europe, vegetation is commonly exposed not only to gaseous phytotoxicants (such as ozone and sulfur dioxide) common to urban (and even some rural) areas but also to "acid rain". Little information is available to evaluate the potential for interaction effects of wet- and dry-deposited pollutants as far as plant response is concerned.

During significant portions of the growing season, the foliage of

Table 2.2 Thresholds for visible injury and growth effects associated with experimental studies of wet deposition of acidic substances (after Jacobson, 1980a,b)

Effect	Species	Threshold[a]	Reference	Remarks
Foliar lesions, decrease in growth	Yellow birch	pH 3.1	Wood and Bormann	Greenhouse (1974)
Foliar aberrations, decrease in growth	Bean	pH 2.5	Ferenbaugh (1976)	Greenhouse
Foliar lesions	Bean, sunflower	pH 3.1	Evans et al. (1977)	Greenhouse
Foliar lesions	Bean	pH 3.2	Shriner (1978a)	Greenhouse
Foliar lesions	Hybrid poplar	pH 3.4	Evans et al. (1978)	Greenhouse
Foliar lesions	Sunflower	pH 3.4	Jacobson and van Leuken (1977)	Greenhouse
Reduced growth	Bean	pH 4.0	Longnecker and Shriner (unpublished)	Greenhouse
Foliar symptoms, no reduced growth	Soybean	pH 3.0	Jacobson (1980b)	Greenhouse
Increased growth, increased/decreased nutrient content	Lettuce	pH 3.0, 3.2	Jacobson (1980b)	Greenhouse (varied with $SO_4^=$ & NO_3^-)
Reduced growth	Pinto bean	pH 3.1	Jacobson (1980b)	Greenhouse
Reduced yield	Pinto bean	pH 2.7		
Reduced growth	Soybean	pH 3.1		
Reduced yield	Soybean	pH 2.5		
Increased yield	Soybean	pH 3.1		
Foliar symptoms	Tomato	pH 3.0	Jacobson (1980b)	Greenhouse
Reduced growth	Tomato	pH 3.0		
Reduced yield	Tomato	pH 3.0		
Reduced quality	Tomato	pH 3.0		
No foliar symptoms, or effects on growth	Soybean	pH 3.1	Irving (1979)	Field
No foliar symptoms, but (a) decreased growth, yield; (b) increased yield	Soybean Soybean Soybean	pH 2.8 pH 2.8 pH 2.8	Jacobson (1980b)	Field, low ozone Field, high ozone Field, low ozone
No effect on growth, yield	Tomato	pH 3.0	Jacobson (1980a)	Field
Reduced quality	Tomato	pH 3.0		Field
Reduced yield	Soybean	pH 4.0	Evans (personal communication)	Field, ambient rain excluded, ozone not measured
Reduced yield	Field corn	pH 4.0	Lee (personal communication)	Field, no ambient rain, low ozone; no effect at pH 3.0

[a] Highest pH to elicit a negative response or lowest pH to elicit a positive response.

vegetation will be wetted by rain, fog, or dew formation in mid-latitude temperate climates. During these periods leaf surfaces become more efficient sinks for the dry deposition of gases. As SO_2 dissolves in the film of moisture on leaf surfaces, Unsworth and Fowler (1976), found that the pH of the dew increased, suggesting the potential for direct acid-type effects on wet vegetation surfaces.

Preliminary work by Shriner (1978b), Irving (1979), and Jacobson et al. (1980) suggests that interactions may occur. Irving (1979), working with field-grown soybeans, found that simulated acid precipitation (pH 3.1) apparently lessened the effect of SO_2 on photosynthetic decrease resulting from multiple exposures to 0.19 ppm of SO_2 during the growing season. Shriner (1978b) observed a significant growth reduction at harvest of plants exposed to 0.15 ppm ozone (4 3-h exposures) in between four weekly exposures to rainfall of pH 4.0. Jacobson, et al. (1980) demonstrated that ozone depresses both growth and yield of soybeans under three different acid rain treatments, but that the depression was greatest with the most acidic rain. The above investigations clearly point to the need for research in this area for two reasons. First, because ozone levels equal to, or greater than, those used in the study are relatively common in most of the areas of the eastern United States where acid deposition is also a problem (Jacobson et al. 1980). Table 2.3 summarizes the land area of potential concern. Second, mechanisms of interaction are currently unknown and should be investigated.

Uncertainty Associated with the Analysis

Recent experiments of Evans and Lee (both as yet unpublished) point toward a significant improvement in our ability to assess acid-rain-induced crop impacts. L. S. Evans (personal communication), working in the field at a site on Long Island, New York, observed that the seed yields of field-grown soybeans exposed twice weekly to simulated acidic rainfalls of pH 4.0, 3.3, and 2.7 were decreased by 10.7%, 16.8%, and 22.9%, respectively, when compared with plants exposed to control rainfalls of pH 5.6. These experiments represent a significant departure from the earlier studies cited above in that all plants were shielded from ambient rainfall through the use of movable covers. J. J. Lee (personal communication), working with field corn in the field in the Willamette Valley of Oregon, also observed significant yield reductions from simulated rainfall treatments at pH 4.0. However, yields at pH 3.0 treatment level were not signfiicantly different from those from pH 5.6 controls.

In both cases, the crops being studied were grown outside of the geographic range of their commercial production. Obvious questions arise relative to the relationship of the response observed to the differences between soil and climatic variables at those sites compared to the corn and soybean belts. Also, in the case of Evans' work, the ambient ozone dose to which the crop was exposed is unknown, making it impossible to eliminate possible interaction effects between the acid rain and gaseous pollutants (Table 2.3). Potential interactions between positive sulfur and nitrogen inputs and negative hydrogen inputs are also a source of uncertainty.

State of the Art in Crop Loss Assessment Due to Acid Rain

Several developments in the area of methodology for the study of acid rain effects on agricultural crops which occurred during 1981 promise to improve evaluations of acid rain impacts on agricultural crops. Most significant of these developments was the principle of ambient rainfall exclusion. Researchers at Brookhaven National Laboratory, Boyce Thompson

Table 2.3. Summary of agricultural and other areas impacted by acid precipitation (>50 kg/ha annual hydrogen ion deposition) and ozone (>60 ppb maximum 7-h daily average, June-September)

	Ozone			
	Low <60 ppb[1] H^+ deposition		High>60 ppb[1] H^+ deposition	
	Low <50 kg/ha	Hi >50 kg/ha	Low <50 kg/ha	Hi >50 kg/ha
Non-agriculture (<30% agricultural)				
Number of counties	462	129	137	41
Area (Ha*1000)	195,000 (25.4%)	19,000 (2.5%)	18,000 (2.4%)	7,000 (0.9%)
Agriculture (>30% agricultural)				
Number of counties	1,346	417	360	142
Area (Ha*1000)	377,000 (49.0%)	54,000 (7.0%)	65,000 (8.5%)	20,000 (2.6%)
Total area	572,000	73,000	83,000	27,000

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[1] Maximum 7-h average, June-September.

Institute for Plant Research, Oak Ridge National Laboratory, and North Carolina State University are currently in the process of using or developing such facilities for crop loss assessment.

Field research with simulated rain (with ambient rainfall excluded) and with ambient gaseous pollutants excluded will assist crop loss assessment relative to acid rain. These combinations of wet and dry pollutant exclusion techniques are necessary to be able to understand the net effect of the combined pollutant exposure.

CONCLUSIONS

Because of the large number of uncertainties associated with experiments and methods up to 1981, it is not currently possible to assess acid rain crop impacts as has been done for ozone. However, through application of new methodology and through the use of coordinated experiments at a group of sites, initiation of such an assessment should be possible within the next five years. The most consistent conclusion to be drawn from experiments done under both natural and laboratory conditions and with the various species has been one of "no effect" at current average ambient pH levels (pH 4.0-4.2). In terms of economic analysis, however, two important exceptions were noted with corn and soybeans (Table 2.2), and the importance of these crops within the region most heavily impacted by acid rain and ozone (Fig. 2.1) requires that further study be given to understanding agricultural impacts of acid rain.

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3. FATE AND EFFECTS OF ACIDIC DEPOSITION IN FOREST ECOSYSTEMS

INTRODUCTION

Upon entering a forest ecosystem, acid rain becomes part of a chemical system regulated by a variety of natural processes. Hydrogen (H^+), sulfate (SO_4^{2-}), and nitrate (NO_3^-) ions are produced and cycled naturally within forest ecosystems -- acid rain should be viewed as an addition to these natural cycles. Furthermore, sulfate and nitrate contain essential plant nutrients (S and N) in available forms.

In the next section, the possible fates and effects of atmospherically deposited H^+, SO_4^{2-}, and NO_3^- in forest nutrient cycles are discussed. The following section examines the fate and effects of H^+, SO_4^{2-}, and NO_3^- deposited in excess of amounts that can be biologically utilized, or that enters the control of physical-chemical components of forest nutrient cycles.

BIOLOGICAL INTERACTIONS OF H^+, SO_4^{2-}, AND NO_3^-

Effects on Forest Cation Nutrient Status

Acid rain (and naturally-produced acids) can replace cation nutrients (such as calcium and magnesium) in forest canopy, litter, or soil layers, and cause the loss of these essential nutrients from the forest ecosystem. However, no cation leaching will occur unless the acid-producing substances in rainfall are mobile in the ecosystem (Johnson and Cole 1980), since solutions entering and leaving the ecosystem must be charge-balanced. Thus, it is very unlikely that atmospheric nitrate inputs will significantly affect leaching in most forests, because nitrogen deficiencies are very common and nitrate is readily taken up by forest vegetation (Abrahamsen 1980c, Raynal et al.

1980). Sulfate, the other major acid-producing substance in rainfall, can travel through many types of forest soils.

There is little doubt that acid rain can accelerate cation leaching rates, but the magnitudes of these increases must be evaluated within the context of natural, internal leaching processes, and are quite variable depending on the amount of acid input, the rate of soil leaching by natural processes (e.g., Cronan et al. 1978, Cole and Johnson 1977, Mollitor and Berg 1980), and the degree to which soils are able to buffer against leaching (e.g., by adsorbing sulfate; Johnson and Cole 1977). Furthermore, the ultimate effects of accelerated cation leaching on forest cation nutrient status depend on a number of variables, most notably on the amount of exchangeable cations, primary mineral weathering rate (Raynal et al. 1980, Stuanes 1980), forest cation nutrient requirement, and management practices such as harvesting (Johnson 1981a).

Table 3.1 compares the effects of internal acid production and tree harvesting on cation nutrient status. The table shows that atmospheric acid inputs vary not only in absolute magnitude, but also in their importance relative to natural leaching processes and effects of harvesting -- both of which also vary quite considerably. At an unpolluted site (Findley Lake), natural leaching processes and base cation removal by harvesting exceed atmospheric H^+ inputs. However, even at a polluted site (Solling, West Germany), internal acid production exceeds atmospheric H^+ inputs as measured by open-bucket collectors. In this case, the comparison is misleading, however, because dry deposition to the forest canopy at Solling is exceedingly high (Ulrich at al. 1980). Consequently, H^+ inputs to the forest floor substantially exceed those deposited above the canopy.

Table 3.1

Atmospheric H^+ inputs vs cation removal by internal H^+ production (carbonic and organic acids) and potential net annual cation removal in bole only and whole-tree harvesting in selected forest ecosystems (adapted from Evans et al., 1981)

Site	Species	Age (year)	Precipitation H^+ input[a]	Cation leaching by internal acid production[b]	Cation removal by harvesting[c]	
					Bole	WTH
			$(eq \cdot ha^{-1} \cdot year^{-1})$			
Thompson,	*Pseudotsuga*	42	240[d]	420[d]	380[e]	660[e]
Washington	*mensiesii*		(4.3)	(5.9)		
Solling,	*Fagus sylvatica*	59	900[f]	1950[f]	220[f]	370[e]
W. Germany						
Jadraas,	*Pinus sylvestris*	-	190[f]	226[f]	-	-
Sweden						
Findley,	*Abies amabilis,*	175	90[g]	1410[f]	272[e]	460[e]
Washington	*Tsuga mertensiana*		(5.6)	(4.5)		
H. J. Andrews,	*Pseudotsuga*	450	28[f]	22700[f]	60[e]	106[e]
Oregon	*menziesii*					

[a] Weighted average $[H^+]$ times precipitation amount; weighted average, $[H^+]$ as pH, appears in parentheses where available.

[b] Calculated from net increase in weighted average HCO_3- or organic anion concentration (the latter estimated by anion deficit) times water amount. Weighted average $[H^+]$ as pH for solutions appears in parentheses where available.

[c] Nutrient content divided by age; WTH = whole tree harvest, removal of all aboveground biomass.

[d] From Cole and Johnson (1977).

[e] From Cole and Rapp (1981).

[f] From Andersson et al. (1980). For comparison in this table, only H_2CO_3 production values are included.

[g] From Johnson (1975) and Johnson et al. (1977).

The importance of accelerated cation leaching to forest nutrient status depends on the magnitude of the cation reserves within that system. If the reserves are large relative to leaching rates (such as in calcareous soils), a doubling or tripling of leaching rates due to acid rain is probably of little consequence. If reserves are small (such as in highly-weathered soils), doubling or tripling of leaching rates may have long-term significance. Cases in which soil reserves can be significantly depleted by acid rain (at current input levels) over less than many decades are rare, however. Nonetheless, given a sufficiently large input for a sufficient amount of time, acid rain must eventually produce decreases in cation status.

Currently, few forests of the United States have cation deficiencies, although notable exceptions are known in certain forests of the Northeast (e.g., Stone and Kszystyniak 1977). An intensive study of the effects of acid rain in a forest of the central Adirondack Mountains of New York indicates that rates of nutrient cation leaching have been substantially increased by atmospheric H_2SO_4 deposition (Mollitor and Raynal 1982). Since soil-available cation reserves are quite low in these soils, the researchers concluded, "Chronic leaching by H_2SO_4 combined with internally generated organic acids may represent a real threat to the nutrient status of many Adirondack forest soils" (Mollitor and Raynal 1982: p. 137). However, the authors acknowledge that total soil cation reserves and weathering rates (or rates at which these reserves become available to trees) are unknown, and that these factors must be included to provide a complete assessment. Acid rain may itself cause increases in soil weathering rates, thus potentially offsetting the accelerated leaching losses of cation nutrients to some extent.

Effects on Aluminum Mobilization

Soil scientists have long known that aluminum (Al) is mobilized from soils under sufficiently acid conditions (Bache 1980, Yuan 1963). Also well-known, is that aluminum in sufficient concentrations is toxic to plant roots, either killing them directly or interfering with nutrient uptake (especially phosphorous and calcium; Ragland and Coleman 1959; Pratt 1965). Recently, Ulrich et al. (1980) and Matzner and Ulrich (1981) showed a marked increase in soil aluminum concentrations over a 13-year period in a beech and Norway spruce forest soil at the Solling site in West Germany. They attributed the change to a combination of internal acidifying processes (most notably nitrification) and atmospheric acid inputs. Furthermore, the authors believe that aluminum levels have become toxic to tree roots (especially because of low calcium/aluminum ration), posing a situation with "serious consequences for forestry in Central Europe."

The findings of Ulrich et al. (1980) and Matzner and Ulrich (1980) are cause for concern, but they may not be applicable to all forest types. Acid and sulfate inputs are very high at the Solling site and must be so to have significantly acidified a soil that was intially very acid. (Ulrich et al. report a pH of 3.3 in $CaCL_2$ solution in surface soils and a pH of 4.0 in subsoils at 50 cm in 1966.) Furthermore, species vary widely in their susceptibility to aluminum toxicity.*

Experiments on Norway spruce and several other species in Norway showed that trees can exhibit a remarkable tolerance to acid inputs and high aluminum levels. After three years of irrigation with pH 5.4, 4.0, 3.0, and 2.5 solutions of H_2SO_4 (with and without liming) on Norway spruce seedlings, no significant effects on growth were noted, although a trend toward increasing growth with increasing acid inputs existed (Ogner and Teigen 1980a). Plants irrigated with pH 2.5 and, to some extent, pH 3 solutions, contained excessively high concentrations of aluminum and sulfur, even with liming. No toxic effects of aluminum were noted.

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* Humphreys and Truman (1964) report no adverse effects of solution aluminum concentrations of up to 20 mg/L on the growth of Pinus radiata seedlings, and McCormick and Steiner (1978) report great variations in toxicity thresholds among several tree species, ranging from 10 mg/L Al in solution for a poplar (Populus) hybrid to 80 to 120 mg/L in oak (Quercus supp.), birch (Betula spp.), and pine (Pinus spp.). Ulrich et al. (1980) report Al concentrations in "equilibrium soil solution" (i.e., solution obtained after extraction of soil with water for 24 h) of less than 3 mg/L. More recently, Matzner and Ulrich (1981) report Al concentrations of nearly 20 mg/L in soil solutions from the spruce stand at Solling.

Recently, a significant decline in the growth of several pine species (Johnson et al. 1981; A. H. Johnson et al. in press; Puckett, in press), chestnut oak (Puckett, in press), and red spruce (Roman and Raynal 1980; Siccama et al., in press; A. H. Johnson et al., in press) has been noted in various parts of the northeastern United States. In the case of red spruce, the growth decline was accompanied by considerable mortality. Climatic patterns did not account for these trends, and some researchers suggested that acid rain may have been a factor (Roman and Raynal 1980). This remains a viable hypothesis in many cases (e.g., the New Jersey pine barrens: Johnson et al. 1981), but recent work by A. H. Johnson et al. (in press) fails to support the soil aluminum hypothesis in the case of red spruce. Specifically, no changes in soil pH over a 15-year period (1965-1980) were noted and no relationship between tree vigor and root or foliar aluminum concentration existed. The causes for the decline in red spruce as well as in the other species mentioned above are unknown as of this writing. Acid rain effects via soil-mediated aluminum response are not evident in the red spruce case, but acid rain (perhaps via other mechanisms) remain one of several working hypotheses being investigated at the present time.

However, it must be emphasized that acid rain effects are site-specific phenomena, depending on the amount of acid input and the vegetation, soils, and nutrient status of the site receiving such inputs. Forest ecosystems in general cannot be expected to respond to acid rain in any single way.

Effects on Forest Sulfur and Nitrogen Status

Sulfate and nitrate contain the essential plant nutrients sulfur and nitrogen in available form. Nitrogen-deficient forests are common through the United States, but sulfur-deficient forests are much less common, occurring primarily in the Pacific Northwest (Youngberg and Dyrness 1966, Will and Youngberg 1978, Turner et al. 1977, 1979).

Plant needs for sulfur and nitrogen are closely coupled, thus the status of sulfur and nitrogen in forest ecosystems must be considered together (Turner and Lambert 1980). Studies in a variety of forest tree species show that the ratio by weight of organic sulfur to organic nitrogen in plant tissues is 0.069. The sulfur present in excess of that dictated by this ratio accumulates and cycles as sulfate (e.g., Turner et al. 1980). One can compare SO_4^{2-}, NO_3^-, and NH_4^+-N deposition rates with the range of tree sulfur and nitrogen uptake values (shown in Table 3.2) to gain a general idea of the degree to which atmospheric inputs could contribute to tree requirements.* Much of the eastern United States appears to receive inputs equal to or in

---

* Because root data are seldom available, these uptake estimates are based on aboveground tree components only and therefore represent minimal estimates of total tree uptake. The few studies available suggest that total tree uptake may be 20 to 50% greater than values derived from aboveground tree components only.

Table 3.2 Average annual uptake of N and calculated organic S in several forest types (N data from Cole and Rapp, 1981)[a]

Forest type (n = No. of stands)	Nitrogen Average	Nitrogen S.D.	Nitrogen Range	Sulfur Average	Sulfur S.D.	Sulfur Range
	(kg•ha^{-1}•year^{-1})					
Boreal Coniferous (n = 3)	5.1	2.2	2.8-6.6	0.4	0.2	0.2-0.5
Boreal Deciduous (n = 1)	25.0	-	-	1.7	-	-
Temperate Coniferous (n - 13)	47.4	17.3	34.0-84.4	3.3	7.2	2.4-5.8
Temperate Deciduous (n = 14)	75.4	18.2	43.0-114.9	5.2	1.3	3.0-7.9

[a] For aboveground biomass only.

excess of tree sulfur requirements (Table 3.2, Fig. 3.1). Some areas of the western United States may receive insufficient sulfur inputs for optimal growth, but the level of resolution in these data is insufficient to detect this.

Atmospheric inputs of NH_4^+ and NO_3^- could contribute substantially to nitrogen nutrition in the relatively less-demanding forests of the North Central and Northeastern states, especially in areas near the eastern Great Lakes (Fig. 3.2). In most areas of the west, atmospheric nitrogen inputs are much less significant.

As described earlier, trees require nitrogen and sulfur in a fairly strict ratio (i.e., a N:S ratio of 14.5 on a weight basis for protein synthesis). The ratio of nitrogen to sulfur in atmospheric deposition is far from this ideal value throughout the United States (Fig. 3.3). Thus, it would appear that sulfur inputs are now, or will become, excessive relative to nitrogen inputs, and benefits of sulfur depostion might best be obtained in forests fertilized or otherwise enriched in nitrogen (e.g., Turner et al. 1979).

Forests recycle nutrients (via litterfall, crownwash, etc.) and are therefore not entirely dependent on atmospheric inputs to meet annual nutrient requirements. When atmospheric nutrient inputs are insufficient to meet growth requirements and soil-available nutrient supplies are low (as is often the case for nitrogen), conservation mechanisms within forest nutrient cycles and translocation mechanisms within trees themselves help supply nutrients to growing tissues. Thus, while increased atmospheric nitrogen and sulfur inputs may be beneficial to forests limited by those nutrients, it should not be implied that atmospheric inputs must equal plant uptake to maintain site

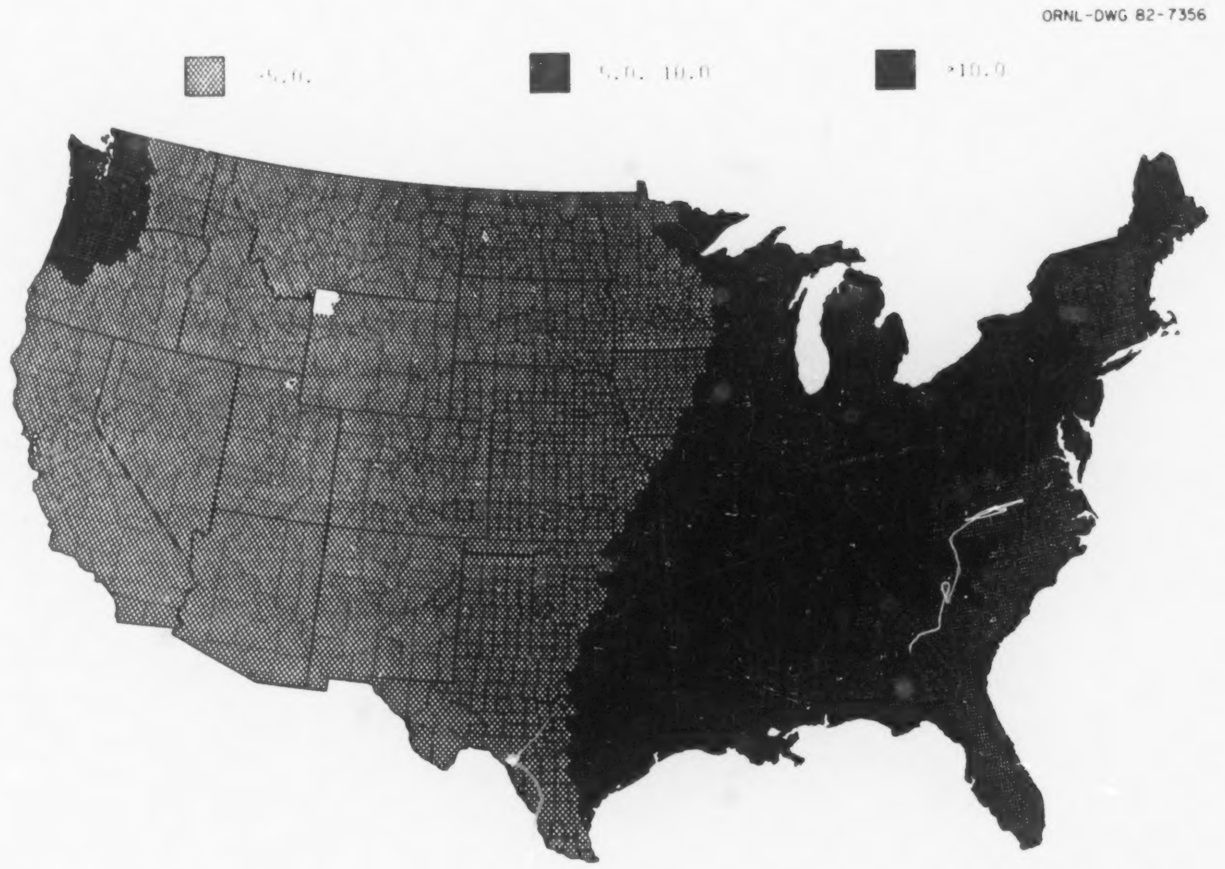
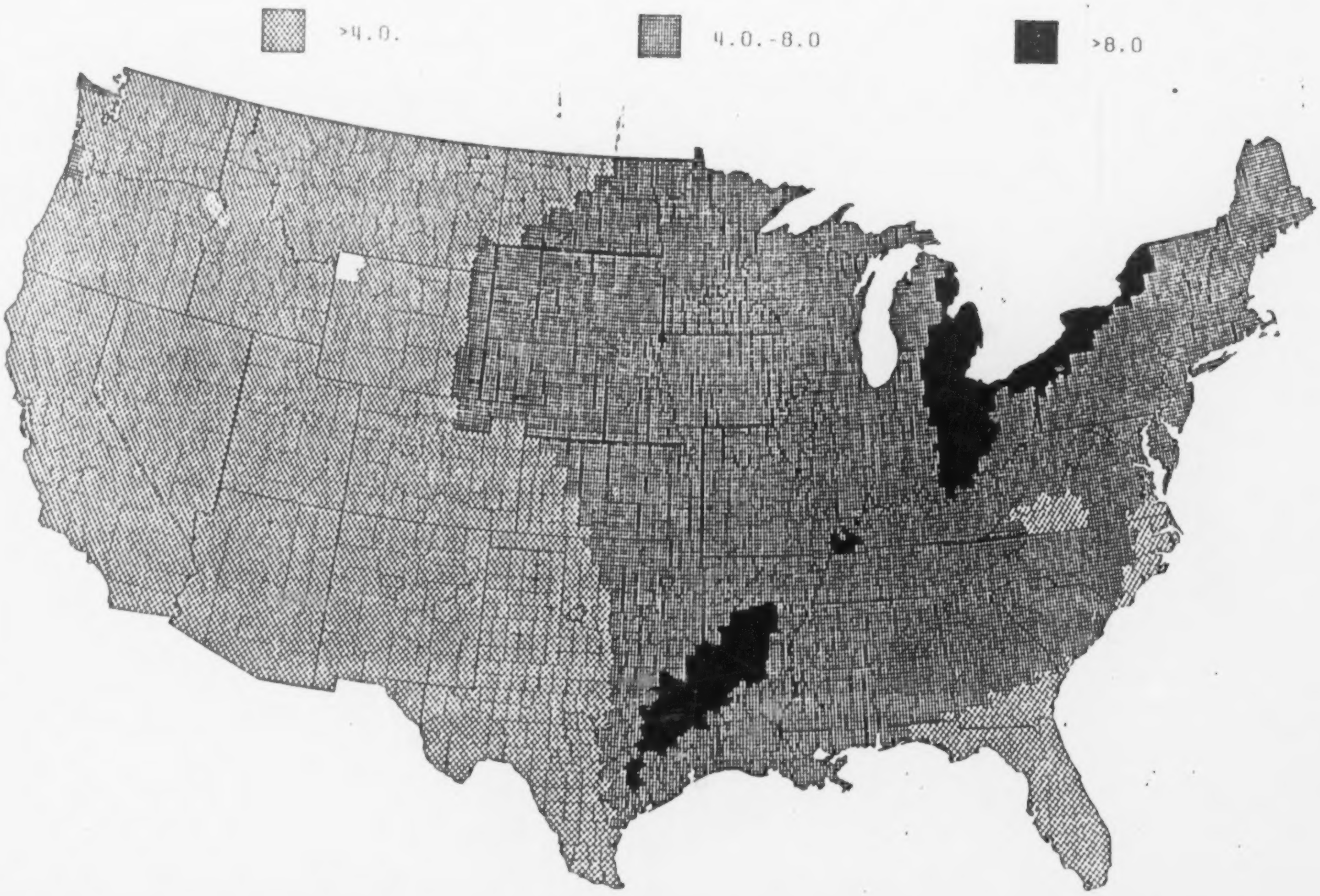
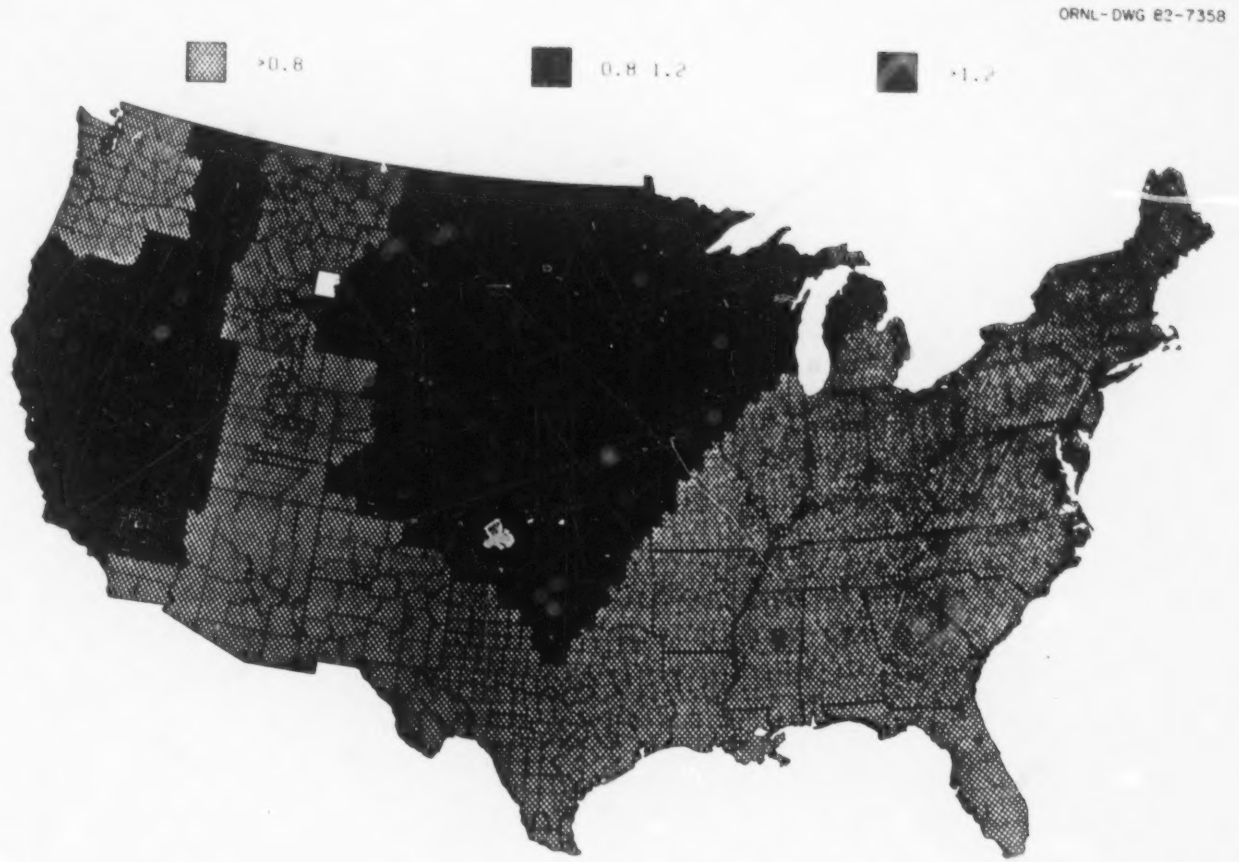

ORNL-DWG 82-7356
<5.0.
5.0- 10.0
>10.0

Fig. 3.1. SO_4-S annual atmospheric deposition (kg/ha) based on the April 1979 to March 1980 period as measured by the National Atmospheric Deposition Program.

Fig. 3.2. $NH_4^+ + NO_3^-$-N annual atmospheric deposition (kg/ha) based on the April 1979 to March 1980 period as measured by the National Atmospheric Deposition Program.

ORNL-DWG 82-7360





ORNL-DWG 82-7358
>0.8
0.8 1.2
>1.2

Fig.3.3 . Ratio of $NH_4^+ + NO_3$-N to SO_4-S in annual atmospheric deposition based on the April 1979 to March 1980 period as measured by the National Atmospheric Deposition Program.

fertility under undisturbed conditions. If the forests in question are subject to harvesting, however, atmospheric inputs must equal export in biomass and subsequent leaching plus runoff to maintain site fertility.

PHYSICAL-CHEMICAL INTERACTIONS OF H^+, SO_4^{2-}, AND NO_3^-

The Anion Mobility Model

Deposition of H^+, SO_4^{2-}, and NO_3^- in excess of amounts that can be biologically utilized within the forest ecosystem will result in one of several possible situations: (1) consumption or entrapment of these ions by chemical-physical processes within the soil or bedrock, (2) exchange of these ions for others within the soil or bedrock, or (3) direct transport of these ions from terrestrial to aquatic ecosystems.

Since solutions entering and leaving a watershed must be charge-balanced, Nye and Greenland (1960) first suggested that a lack of mobile anions (such as sulfate and nitrate) in forest soils limits cation leaching. Seip (1980) concluded that the contributions of direct input and soil pH change are small and therefore the introduction of sulfate (which appears to be relatively mobile in these ecosystems) is the most likely explanation for surface water acidification in Norway. Because soils in these regions are acidic with low base contents, a significant portion of the increased cation concentration required to balance the increased sulfate concentration must be H^+ and Al^{3+}. Empirical studies support this theory (Abrahamsen and Stuanes 1980), but there remains some controversy about the magnitude of changes in surface water acidity this mechanism could produce relative to observed pH changes.

Sulfate mobility can be strongly affected by chemical processes of adsorption in many forest soils (Johnson and Cole 1977; Johnson et al. 1979, 1980; Singh et al. 1980; Mollitor and Berg 1980). Free iron and aluminum hydrous oxides are known to adsorb sulfates (Harwad and Riesenauer 1966), but recent studies showed exceptions to this general rule in some soils.

Nitrate is immobilized in most forest soils because nitrogen is commonly a limiting nutrient in forest ecosystems (Abrahamsen 1980c, Raynal et al. 1980). Therefore, it is unlikely that atmospheric HNO_3 inputs will cause a significant amount of soil leaching or surface water acidification (if incoming rain passes through the soil). Furthermore, nitrate uptake by plants requires current H^+ uptake or exudation of OH^- or HCO_3^-, either of which has a basifying effect on soil solution (Nye 1981).

Sensitivity Criteria

Several sets of sensitivity criteria have been proposed and used to define geographical regions most susceptible to acid rain effects. Each set of criteria is based on different scientific concepts and is aimed at different target organisms or ecosystems (e.g., forests vs. fish, terrestrial vs. aquatic ecosystems). Those directed toward aquatic effects emphasized bedrock geology (Norton 1980, Hendry et al. 1980), while those directed toward soils effects emphasized cation exchange capacity (CEC) and base saturation (McFee 1980, Klopatek et al. 1980).

Table 3.3 and Figs. 3.4 and 3.5 depict hypothetical sensitivies of terrestrial and aquatic ecosystems to the combined effects of CEC, pH, and sulfate adsorption capacity. For non-sulfate-adsorbing soils (Fig. 3.4), it is assumed that each equivalent of incoming H^+ causes the leaching of an equivalent of some cation (including H^+ itself or Al^{3+}) through the forest soil.

Table 3.3 Theoretical sensitivities of terrestrial and aquatic ecosystems to acid precipitation effects given soil base content (eq/ha) and soil sulfate adsorption capacity (eq/ha) (adapted from Johnson, 1981b)

	Soil properties		Terrestrial sensitivity to:			Aquatic sensitivity[a]		
	pH	CEC	Base cation loss	Soil acidification	Al solubilization	Base cation input	Acidification	Al input
				I. Non-sulfate-adsorbing soils				
1.	>6	High	High	Low	Low	High	Low	Low
2.	>6	Low	High	Low-moderate	Low-moderate	High	Low-moderate	Low-moderate
3.	5-6	High	High	Moderate	Moderate	High	Moderate	Moderate
4.	5-6	Low	High	High	Moderate	High	Moderate	Moderate
5.	<5	High	Moderate	Moderate	Moderate	Moderate	High	Moderate
6.	<5	Low	Moderate	Moderate	High	Moderate	High	High
				II. Sulfate-adsorbing soils				
7.	>6	High	Moderate	Low	Low	Moderate	Low	Low
8.	>6	Low	Moderate	Moderate	Low-moderate	Moderate	Low	Low
9.	5-6	High	Low	Moderate	Moderate	Low	Low	Low
10.	5-6	Low	Low	High	Moderate-high	Low	Low	Low
11.	<5	High	Low	Moderate	Moderate-high	Low	Low	Low
12.	<5	Low	Low	High	High	Low	Low	Low

[a] Assuming bedrock geology is sensitive.

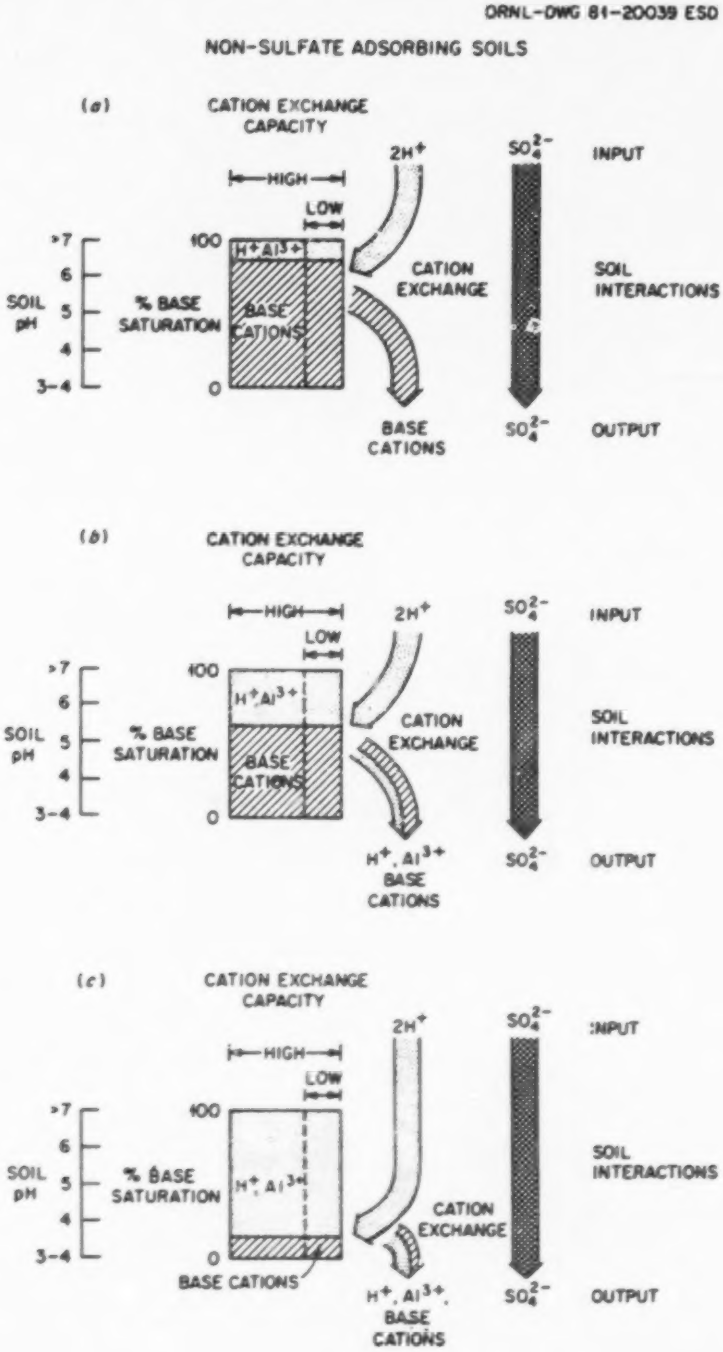

ORNL-DWG 81-20039 ESD
NON-SULFATE ADSORBING SOILS
(a)
CATION EXCHANGE CAPACITY
HIGH
LOW
100
0
H⁺, Al³⁺
BASE CATIONS
% BASE SATURATION
SOIL pH
>7
6
5
4
3-4
2H⁺
CATION EXCHANGE
BASE CATIONS
SO₄²⁻
INPUT
SOIL INTERACTIONS
SO₄²⁻
OUTPUT
(b)
CATION EXCHANGE CAPACITY
HIGH
LOW
100
0
H⁺, Al³⁺
BASE CATIONS
% BASE SATURATION
SOIL pH
>7
6
5
4
3-4
2H⁺
CATION EXCHANGE
H⁺, Al³⁺ BASE CATIONS
SO₄²⁻
INPUT
SOIL INTERACTIONS
SO₄²⁻
OUTPUT
(c)
CATION EXCHANGE CAPACITY
HIGH
LOW
100
0
H⁺, Al³⁺
BASE CATIONS
% BASE SATURATION
SOIL pH
>7
6
5
4
3-4
2H⁺
CATION EXCHANGE
H⁺, Al³⁺, BASE CATIONS
SO₄²⁻
:NPUT
SOIL INTERACTIONS
SO₄²⁻
OUTPUT

Fig. 3.4 Schematic diagrams of soil leaching in non-sulfate-adsorbing soils.

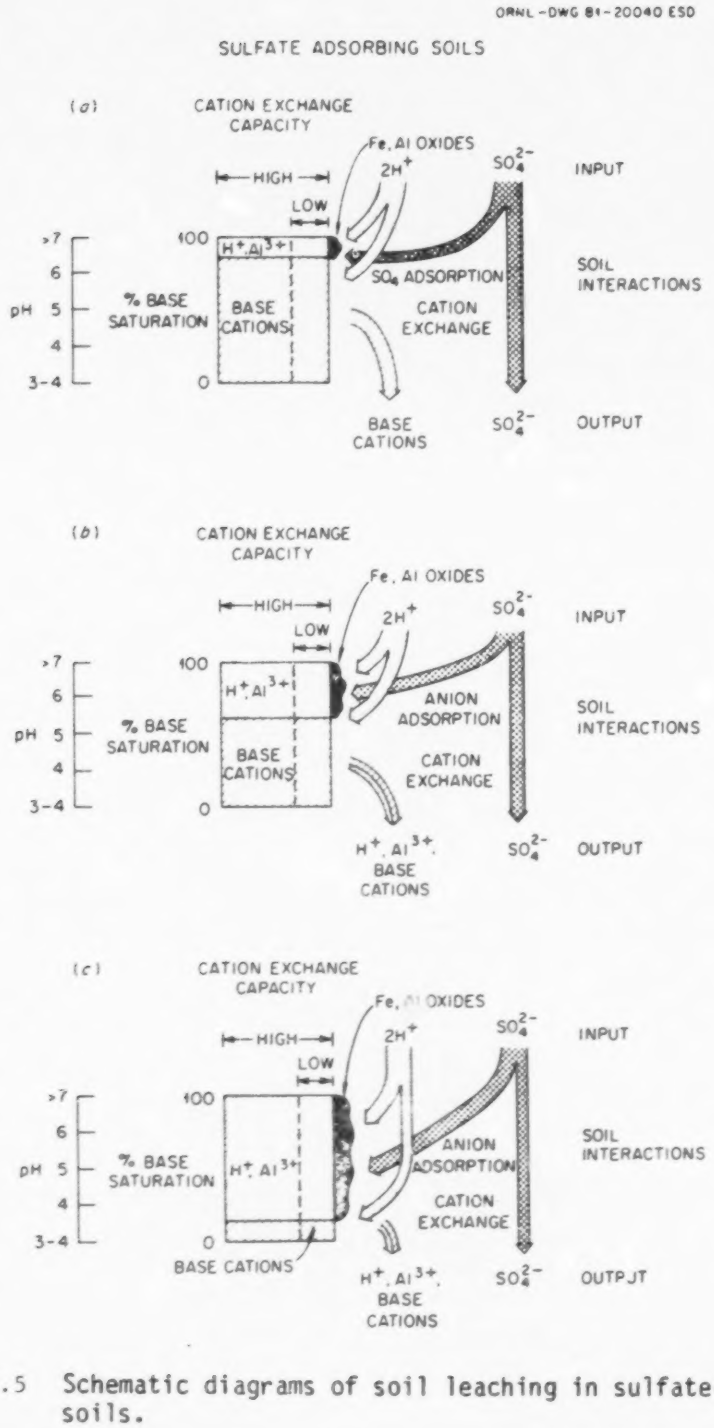

ORNL-DWG 81-20040 ESD
SULFATE ADSORBING SOILS
(a)
CATION EXCHANGE CAPACITY
HIGH
LOW
Fe, Al OXIDES
2H+
SO4 2-
INPUT
>7
6
pH 5
4
3-4
100
0
% BASE SATURATION
H+, Al3+
BASE CATIONS
SO4 ADSORPTION
CATION EXCHANGE
SOIL INTERACTIONS
BASE CATIONS
SO4 2-
OUTPUT
(b)
CATION EXCHANGE CAPACITY
HIGH
LOW
Fe, Al OXIDES
2H+
SO4 2-
INPUT
>7
6
pH 5
4
3-4
100
0
% BASE SATURATION
H+, Al3+
BASE CATIONS
ANION ADSORPTION
CATION EXCHANGE
SOIL INTERACTIONS
H+, Al3+, BASE CATIONS
SO4 2-
OUTPUT
(c)
CATION EXCHANGE CAPACITY
HIGH
LOW
Fe, Al OXIDES
2H+
SO4 2-
INPUT
>7
6
pH 5
4
3-4
100
0
% BASE SATURATION
H+, Al3+
BASE CATIONS
ANION ADSORPTION
CATION EXCHANGE
SOIL INTERACTIONS
H+, Al3+, BASE CATIONS
SO4 2-
OUTPUT

Fig. 3.5 Schematic diagrams of soil leaching in sulfate-adsorbing soils.

Case 1. For soils with pH > 6, H^+-base cation exchange is likely to be nearly 100% efficient (Wiklander 1974, 1980b). Thus soils are very "sensitive" to base cation loss, and aquatic ecosystems will receive additional base cation inputs (balanced by SO_4^{2-}) but not H^+ and Al^{3+} (Fig. 3.4a). If the soil with ph > 6 has a high CEC (i.e., a large reserve of exchangeable cations and large buffering capacity), it will take a very long time for a given acid input to acidify it. This is depicted by the width of the CEC box in Fig. 3.4a. Thus, such soils and adjacent aquatic ecosystems are thought to have low sensitivity to acidification and aluminum mobilization.

Case 2. If the soil with pH > 6 has a low CEC (area depicted to the right of the dashed line in the CEC box in Fig. 3.4a), it will take less time to deplete the exchangeable cation reserves and, therefore, a low-moderate rating is (arbitrarily) assigned to acidification and aluminum mobilization in both soils and aquatic ecosystems to differentiate it from case 1. As in case 1, H^+-cation exchange is nearly 100% complete, so that soils are "sensitive" to base cation loss and aquatic systems receive base cation inputs.

Case 3. If a soil has pH 5-6 (i.e., a moderate base saturation), H^+-cation exchange will be nearly as complete as in cases 1 and 2, while cation reserves (at a given CEC) will be lower (Fig. 3.4b). For the high-CEC case, a moderate rating is assigned to acidification and aluminum mobilization in terrestrial and aquatic ecosystems. As in cases 1 and 2, soils are "sensitive" to base cation loss and aquatic systems (initially) receive base cation inputs.

Case 4. In this case, the total reserves of base cations are low, yet H^+-cation exchange is nearly 100% efficient (Fig. 3.4b) -- thus the soil is highly sensitive to base cation loss and acidification. Once base cations are depleted and the soil is acidified, aluminum may become mobilized; thus, a moderate rating is assigned to soil aluminum mobilization. Aquatic systems receiving waters from such soils will first receive additional base cation inputs and then H^+ and AL^{3+} as soils become acid. Thus, aquatic ecosystems are sensitive to acidification and Al^{3+} input, but not immediately, and a moderate rating is assigned.

Case 5. In soils with pH < 5 (i.e., low base saturation), H^+-base cation exchange is less efficient; therefore, soils are only moderately sensitive to base cation loss and further acidification (Fig. 3.4c). Such a soil may be sensitive to aluminum mobilization given sufficiently high acid inputs, however. In the high-CEC case, a moderate sensitivity is assigned to aluminum mobilization in soils and aquatic ecosystems. Aquatic ecosystems in case 5 are sensitive to acid inputs according to Seip's (1980) hypothesis, but will receive additional base cation inputs also.

Case 6. In this case, the soil is acid and has low-CEC (Fig. 3.4c), making it only moderately sensitive to base cation loss and further acidification, but highly sensitive to aluminum mobilization. Aquatic ecosystems receive additional base cation inputs but, according to Seip's hypothesis, are highly sensitive to acidification and Al^{3+} input.

In sulfate-adsorbing soils, leaching of H^+, Al^{3+}, and base cations is inhibited for the reasons described previously. The degree of sulfate adsorption is dependent to some extent on pH (Harward and Reisenauer 1966), as well as on inherent soil properties such as organic matter and Fe- and Al-oxide content. Sulfate is more strongly adsorbed in most acid soils, thus hypothetically rendering aquatic ecosystems insensitive to H^+ and Al^{3+} inputs. At higher pH's where sulfate adsorption is not as strong, aquatic ecosystems are subject to base cation inputs (to a lesser extent than in non-sulfate-adsorbing soils), but if soils acidify, sulfate adsorption increases and reduces leaching. This negative feedback mechanism theoretically makes aquatic ecosystems uniformly insensitve to acidification and Al^{3+} inputs where soils adsorb sulfate (Table 3.3, Fig. 3.5a-c).

Cases 7 to 12 are analogous to cases 1 to 6, respectively, with regard to pH and CEC. Due to the sulfate adsorption capacity of soils in cases 7 to 12, however, base cation leaching is lower than in the analogous soils in cases 1 to 6. However, sulfate adsorption prevents H^+ transport as well as base cation leaching from soils in cases 7 to 12, and the displacing efficiency of H^+ for base cations is less of a factor in soil acidification. Thus, soils in cases 7 to 12 are rated more sensitive to acidification and aluminum solubilization than their counterparts in cases 1 to 6.

The preceding discussion, as well as Table 3.3 and Figs. 3.4 and 3.5, by no means represent a rigid set of criteria for site sensitivity to acid rain. They merely represent a hypothetical construct based on a combination of known and previously employed sensitivity criteria. These criteria have limitations and are by no means complete. For instance, while considering the sensitivity of terrestrial and aquatic ecosystems to acid preciptation effects, it is important to realize that acids are produced naturally within these systems (Rosenquist 1977). The effects of atmospheric acid inputs must be viewed as an addition to natural, ongoing acidification and leaching processes within soils due to carbonic acid formation, organic acid formation, tree cation uptake, and a variety of other processes (Ulrich 1980, Sollins et al. 1980, Andersson et al. 1980). Unfortunately, the data base for including natural acid formation criteria into regional sensitivity assessments is extremely limited. Thus, current sensitivity rating schemes, by default, assume that atmospheric inputs add significantly to internal acid production, an assumption that is by no means universally accepted (Rosenquist 1977).

Regional Location and Extent of Areas Sensitive to Acidic Deposition

Two maps of regions thought to be sensitive to acid deposition are presented as Figs. 3.6 and 3.7. The first displays the potential for acidification of non-managed soils; the second the potential for acid runoff to lakes and streams. Surface soil properties used in the analysis included exchangeable bases, pH, cation exchange capacity, clay content, base saturation, and sulfate adsorption capacity. Sulfate adsorption was included because it has been shown to prevent leaching of nutrient cations or the transport of acid to aquatic ecosystems (Johnson 1980, Seip 1980). However, this analysis presents a minimal picture of the actual geographic extent of soils possessing sulfate adsorption capacity due to data limitations. Bedrock geology in terms of potential to buffer acid was included. Topographic characteristics used were steepness (relief) and elevation. Land use was used to distinguish managed areas, such as crop land and urban areas, from forest and range areas. [See Olson et al. (1982) for full details.]

County-level data on soils, land use, and terrain characteristics were obtained from the Geoecology Data Base (Olson et al. 1980). Data on bedrock sensitivity were obtained from the Brookhaven National Laboratory ACID data

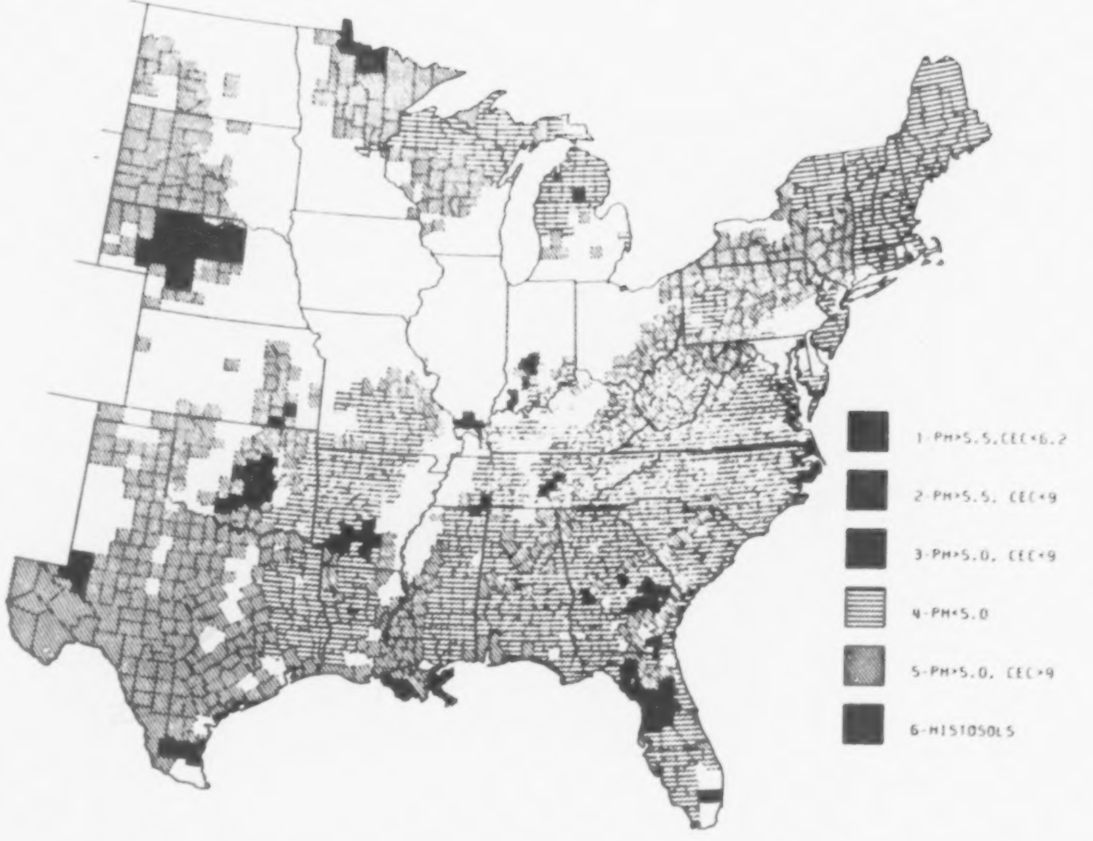

1-PH>5.5, CEC<6.2
2-PH>5.5, CEC<9
3-PH>5.0, CEC<9
4-PH<5.0
5-PH>5.0, CEC>9
6-HISTOSOLS

Fig. 3.6. Soils characterisitics that influence the potential acidification of nonmanaged soils (<50%) 50% coverage used to classify counties soils data from 1970 National Atlas.

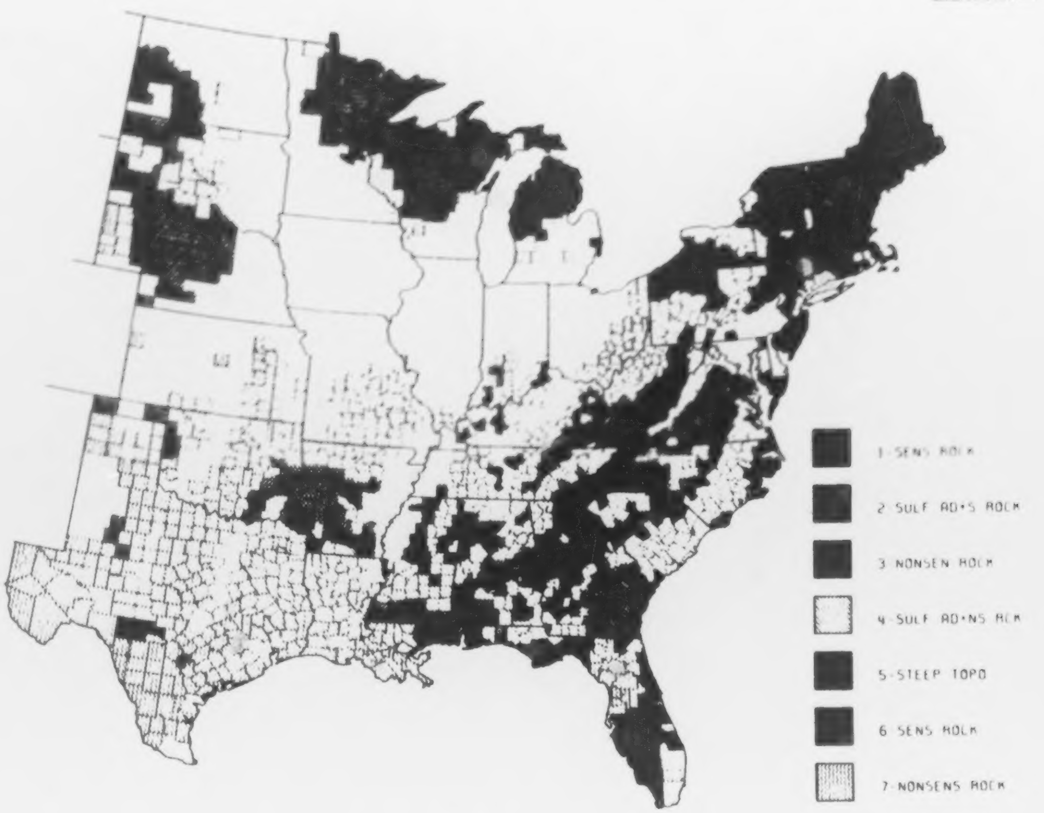

1-SENS ROCK
2 SULF AD+S ROCK
3 NONSEN ROCK
4-SULF AD+NS RCK
5-STEEP TOPO
6 SENS ROCK
7-NONSENS ROCK

Fig. 3.7 Terrain, soils, and bedrock characteristics that influence the potential transfer of acid precipitation from nonmanaged soils (>50%) to watersheds class-1-4: Steep: class 6-7: pH > 5, 50% used to classify counties soils, landform data from 1970 National Atlas, and bedrock data from Norton.

base (Hendrey et al. 1980) and incorporated into the Geoecology Data Base. The Geoecology Data Base contains extensive environmental data for county units in the coterminous United States. It was developed at Oak Ridge National Laboratory to conduct regional studies and is designed for integration of various thematic parameters through the use of common spatial cells (counties).

Six classes were used to characterize soils in the east (Table 3.4), with agricultural and urban areas blank on the sensitivity map (Fig. 3.6). Classes 1 to 3 are specifically related to the increasing potential for soils to undergo acidification with acid deposition. Only one soil type had both moderate pH (6.5) and a low CEC (2.3). This soil type only occurs extensively in 15 counties in Nebraska. It is primarily rangeland and is currently receiving low levels of acid deposition.

Due to the level of resolution of mapping and the soils information within the Geoecology Data Base, it is likely that some soils fitting the Wiklander criteria do exist within areas receiving high levels of acid deposition [i.e., the soils of the Adirondacks studied by Raynal et al. (1980)]. However, this exercise suggests that such soils are rare, at best -- thus according to the Wiklander (1974, 1980) criteria, using the CEC value of McFee (1980), no soils occurring extensively in the eastern United States are sensitive to acidification by acid rain.

Low soil pH was used to classify soils with the lowest potential to reduce the acidity of rainfall. According to Seip (1980), as soil pH decreases below 5.0 there is an increasing probability for streams and lakes within a watershed to receive anion inputs from the terrestrial system from increased sulfate deposition associated with acid rain. Because these soils

Table 3-4 Soil characteristic classification (Olson et al., 1982)

Class	Acidif-potential	No. of counties	Characteristics
1	Hign	16	Moderate pH (>5.5), lowest CEC
(<6.2)			
2	Moderate	19,	Moderate pH (>5.5), low CED (<9.0)
3	Mod-low	45	Moderate pH (>5.0), low CED (<9.0)
4	Low	849	Low pH (<5.0)
5	Low	700	Moderate pH (>5.0), low CED (>9.0)
6	Low	13	Histosols

generally have low base contents, a significant portion of the increased cation concentration required to balance the increased sulfate input must be H^+ and Al^{3+}.

These soils would also have the greatest aluminum mobility with increasing acid inputs. In addition, since these soils are naturally acid, they are the ones most likely to contribute natural acidity to surface waters (Rosenquist 1977). Such soils do indeed have the lowest potential to reduce acidity of rainfall, but they also have the potential to acidify rainfall in areas where no acid deposition occurs (Johnson et al. 1977, Johnson 1981c).

Seven classes were used to describe the combinations of soil, bedrock, and terrain that occurred in the east (Table 3.5) with agriculture and urban areas blank on the sensitivity map (Fig. 3.7). Table 3.6 defines the factors in each class and the assignment of low, moderate, or high potential to reduce acidity of acid inputs.

High potential areas would generally be able to reduce acidity so that impacts to aquatic systems would generally be minimal. The moderate class would be between the extremes in reducing acidity and may also be more variable -- within a moderate county, both low and high areas may occur.

Eight percent of the 2660 counties in the east have low potential to reduce acidity of acid inputs, while an additional 20% have moderate potential (Table 3.6). Classes 3, 4, and 6 are combinations of soils and bedrock having opposing potentials for reducing acidity, i.e., soils having low potential but underlying bedrock having high potential. In these areas, the soil depth and other terrain characteristics (such as glaciation or soil pans) determine whether the soil or bedrock properties will dominate. Class 3 consists of

Table 3.5 Soils with CEC < 9.0 and pH > 5.0 occurring in the Eastern United States (Olson et al., 1982)

Soil	CEC (meq/100 g)	pH	Area (ha x 1000)	Land use Agric. (%)	Land use Forest/range (%)
E1301	2.3	6.5*	7,400	33	63
U0401	5.9	5.0	2,000	33	58
E1302	6.5	6.7*	1,000	22	72
A0601	7.0	5.2	200	46	46
A0607	7.2	5.2	1,700	59	25
E1003	8.0	5.3	800	29	66
N1101	8.0	5.8*	3,500	30	56
A0905	8.7	5.6*	1,700	22	68
A0608	8.6	5.4	700	64	19
E1001	8.7	5.2	1,400	26	66
E1002	8.7	5.2	1,000	19	65
A0609	9.0	5.2	1,000	51	24

Table 3.6 Characteristics of map classes for eastern United States as to the potential to reduce acidity of acid deposition (Olson et al., 1982).

Class	Potential	No. of Counties	Characteristics
1	Low	72	Steep slopes, high relief, high elevation
2	Low	89	Low soil pH, sensitive bedrock
3	Low-High	114	Low soil pH, nonsensitive bedrock
4	Moderate	291	Low soil pH, sensitive bedrock, sulfate adsorption
5	High	326	Low soil pH, nonsensitive bedrock, sulfate adsorption
6	Moderate	241	High soil pH, sensitive bedrock
7	High	515	High soil pH, nonsensitive bedrock

low-pH soils overlaying bedrock with high buffering capacity. In the south these soils are generally very thick, and bedrock would be an insignificant factor. However, in northern glaciated areas, the thin, porous soils would result in a high potential to reduce acidity of precipitation through the interaction with bedrock. The southern limit of the Wisconsin glaciation is shown on Fig. 3.7 to help interpret Class 3 areas.

The most serious deficiency in this and other sets of sensitivity criteria to date is the lack of information on external atmospheric acid input and natural internal production. A site cannot be regarded as sensitive to acid precipitation effects if atmospheric input is insignificant relative to natural acid production (e.g., Rosenquist 1977). Unfortunately, data bases for natural acid production are very limited in geographical extent, making regionl assessments infeasible. Therefore, sensitivity ratings presented here represent sensitivity, given that atmospheric acid inputs will either currently, or in the future, exceed or significantly add to natural internal acid production.

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4. EFFECTS OF OZONE ON FOREST PRODUCTIVITY

4.1 INTRODUCTION

Vast areas of the forested landscape of the United States fall within the regions most frequently experiencing elevated levels of long-range transported air pollutants. The value of this forest resource is large, and growth impacts resulting from air pollutant stress, even if small on a percentage basis, are likely to be significant. However, quantification of such an impact is very difficult because: (1) dose-response information for growth parameters is practically nonexistent, and virtually all is for seedling growth stages; (2) resource inventory data are fragmentary, not accessible in any single data base, or in any single format; (3) management decisions regulating the rotation age, and hence the cumulative dose experienced by a species, vary significantly from species to species and from region to region within a species; and (4) competition between forest tree species in mixed forest stands may alter the individual species response to a stress such as air pollution which may differentially affect the competing species. For these reasons, our analyses of ozone impacts to forest productivity are restricted to a discussion of (1) the state of the art relative to forest species sensitivity, (2) the relationship between the distribution of the forest species and the occurrence of elevated ozone concentrations, and (3) a single "case study" involving the extrapolation of the only growth-related dose response data available (for seedlings) to mature forest stand growth for a sensitive species (eastern white pine, Pinus strobus) important to the forest economy of a 21-county region of Western North Carolina.

4.2 SOURCES OF AVAILABLE DATA

4.2.1 The Forests and Forest Ecosystems of the United States

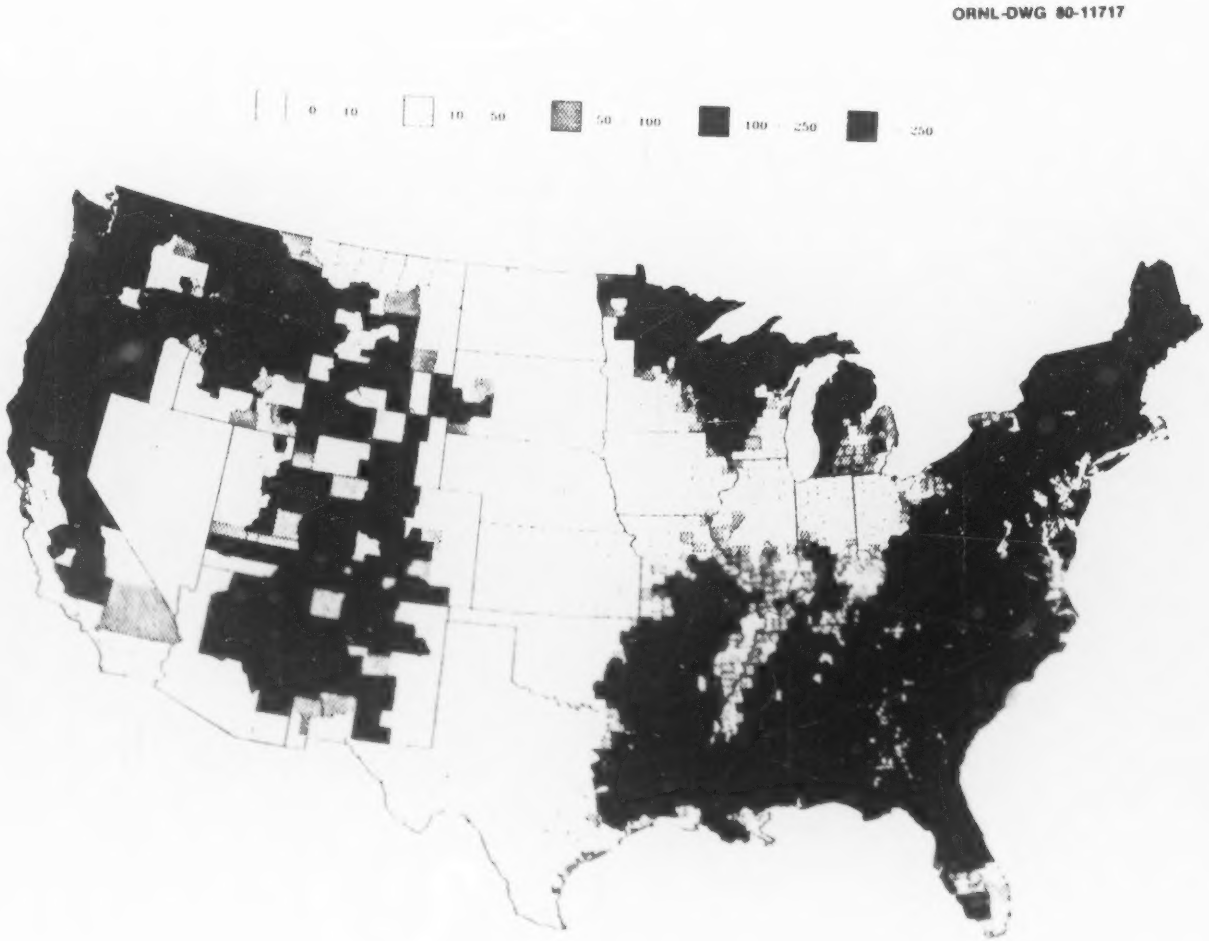
One-third of the total land area of the United States is forested -- two-thirds of that area is classed as commerical timberland. Nearly three-quarters of the 400 million acres of commercial timberland is located in the eastern half of the United States, with about equal distribution between the Northern and Southern sections (Fig. 4.1). Nearly 80% of New England is forested and greater than 40% of the Atlantic Coast states are forested.

Forests consist of numerous species of trees and shrubs and other forms of vegetation such as forbs, grasses, and woody perennials. In such an ecosystem, minor change in one organism may eventually lead to larger changes in community structure and diversity, and may further result in change of entire ecosystems.

4.2.2 Air-Pollutant-Induced Stress to Individual Forest Tree Species

The large majority of studies of ozone effects conducted to date have dealt with the response of individual tree species rather than with the response of forests or forest ecosystems. Only one major study exists in which a substantial effort was made to examine effects to specific portions of a forest system (San Bernardino Mountain Studies; Miller and McBride 1974).

The influence of air pollutants on plant health is a complex process that involves not only pollutant concentration and duration of exposure but also environmental factors that influence the overall response of the plant as a living organism under stress. As with all plant-stress-inducing agents, air pollutants may initiate changes within plant metabolic systems that may lead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0 10
10 50
50 100
100 250
250

Fig. 4.1 Commercial Forest Acreage by County (thousands of acres).

to extensive physiological modifications, and if sufficient change occurs, visible symptoms may be manifested. In some instances, the addition of low pollutant concentrations to a plant's environment may induce a fertilizer-like response. A few studies of this phenomenon on agronomic crops have been completed to date; no studies have shown beneficial effects on natural ecosystems.

Tree death may result from continual exposure to low or high pollutant doses. Other mitigating factors may also be involved (i.e., abiotic or biotic disease-inducing agents or insect attack). Depending on the tree species, the seasonal stage of growth, pollutant dose, and environmental conditions, many forms of injury may take place and their relative impacts may vary. Symptoms of acute and chronic injury may occur on a given plant simultaneously.

Injury does not necessarily imply damage (i.e., economic loss). Timing of pollutant exposure in relation to the stage of development often determines the relationship of visible foliar leaf injury to subsequent yield losses.

In summary, there are several possible responses to air pollutants (Smith 1980a): (1) fertilizer effects appearing as increased growth and yields, (2) no detectable responses, (3) injury manifested as growth and yield reductions without visible symptoms on the foliage, or with very mild foliar symptoms difficult to perceive as air-pollution induced, (4) injury exhibited as chronic or acute symptoms on foliage with or without associated reductions in growth and yield, and (5) death of trees and forest communities.

4.2.3 Response Of Forest Vegetation To Ozone

Considerable literature is available on the effects of O_3 on native vegetation, and much of this literature is summarized in Table 4.1. Effects

Table 4.1 Effects of O_3 on growth, physiology, and foliar injury to woody native vegetation

Species	O_3 concentration (ppm)	Exposure duration	Description of response compared to controls	Ref (see key)
Eastern white pine (*Pinus strobus*)	0.05	6 h	No foliar injury to either sensitive or resistant clones	8
Eastern white pine (*Pinus strobus*)	0.05	4,9,17, 22 weeks continuous	Soluble sugars in primary needles, significantly higher[c] than controls; ascorbic acid also higher[c]	1
Short leaf pine (*Pinus echinata*)	0.05	11 weeks continuous	Significant increase[c] in soluble sugars and ascorbic acid of primary needles	1
Slash pine (*Pinus elliotti*)	0.05	11 weeks continuous	Significant increase[c] in soluble sugars and ascorbic acid of primary needles	1
Loblolly pine (*Pinus taeda*)	0.05	11 weeks continuous	Significant increase[c] in soluble sugars and ascorbic acid of primary needles	1
Pond pine (*Pinus serotina*)	0.05	11 weeks continuous	Significant increase[c] in soluble sugars and ascorbic acid of primary needles	1
Loblolly pine (*Pinus taeda*)	0.05/0.10/0.14	6 h/d, 28 d	5% decrease in height growth due to O_3 alone, 22% decrease[a] due to $O_3 + SO_2$, and 30% decrease[c] due to $O_3 + SO_2 + NO_2$	1
American sycamore (*Platanus occidentalis*)	0.05/0.10/0.14	6 h/d, 28 d	8% decrease in height growth due to O_3 alone, 32% decrease[c] to $O_3 + SO_2$, and 45% decrease[c] due to $O_3 + SO_2 + NO_2$ with no foliar injury	13
Virginia pine (*Pinus virginiana*)	0.07-0.10	10 h/d during two growing seasons	Tracheid length reduced[c]	18
Jack pine (*Pinus banksiana*)	0.07-0.10	10 h/d during two growing seasons	Tracheid length unaffected	18
Ponderosa pine (*Pinus ponderosa*)	0.09 (ambient air)	Average daily peak concentration for 16-d period	Decrease in chlorophyll content of needles	15
White ash (*Fraxinus americana*)	0.10	5 h	Level at which foliar injury (surface strippling) of seedlings occurred	19
Eastern white pine (*Pinus strobus*)	0.10	6 h	No foliar injury to either sensitive or resistant clones	8
Sugar maple (*Acer sacharum*)	0.10	2 h/d for 14 d	Level at which foliar injury began to occur	7
American sycamore	0.10	8 h	Maintained constant apparent photosynthetic rate; no effects on the internal structure of the leaf	14
White ash (*Fraxinus americana*)	0.10	2, 4, 6, 8 h	Foliar injury	22
Hybrid poplar (*Poplar maximowiczii*)	0.10	2, 4, 6, 8 h	Foliar injury	22
Tulip-poplar (*Liriodandron tulipfera*)	0.10	2, 4, 6, 8 h	Foliar injury	22
Austrian pine (*Pinus nigra*)	0.10	8 h	Injury to current year needles	
Jack pine (*Pinus banksiana*)	0.10	8 h	Injury to current year needles	
Virginia pine (*Pinus virgiiana*)	0.10	8 h	Injury to current year needles	
Eastern white pine (*Pinus strobus*)	0.10	7, 21 d	(Pre-exposed) CO_2^{14} fixation reduced	21
Douglas-fir (*Pseudotsuga menziesii*)	0.10	20 weeks; 6 h/d; 7 d/week	Growth reduction in root length (7.5%)[b]; plant height (6.1%)[b]; dry weight of foliage (12.1%)[b]; stem (14.8%)[b]; roots (9.7%)[b]; foliar injury	20
Jeffrey pine (*Pinus jeffreyi*)	0.10	20 weeks; 6 h/d; 7 d/week	Growth reduction in plant height; (2.2%)[b] dry weight of foliage (5.6%)[b]; stem (1.6%)[b]; root (5.4%)[b]; foliar injury	20
Lodgepole pine (*Pinus contorta*)	0.10	20 weeks; 6 h/d; 7 d/week	Growth reduction in plant height (6.6%)[b]; root length (13.0%)[b]; dry weight of foliage (2.4%)[b]; stem (8.4%)[b]; root (14.7%)[b]; foliar injury	20

Table 4.1 (Continued)

Species	O_3 concentration (ppm)	Exposure duration	Description of response compared to controls	Ref (see key)
Monterey pine (Pinus radiata)	0.10	20 weeks; 6 h/d; 7 d/week	Growth reduction in root length (1.6%)^{b}; no corresponding reductions in dry weights or plant height; no foliar injury	20
Ponderosa pine (Pinus ponderosa)	0.10	20 weeks; 6 h/d; 7 d/week	Growth reduction in root length (12.1%)^{b}; plant height (11.3%)^{b}; dry weight of the foliage (12.4%)^{b}; stem (21.3%)^{b}; root (26.3%)^{a}; foliar injury	20
Shore pine (Pinus contorta)	0.10	20 weeks; 6 h/d; 7 d/week	Growth reduction in root length (2.8%)^{b}; plant height (2.2%)^{b}; dry weight of stem (6.3%)^{b}; root (10.1%); foliar injury	20
Sugar pine (Pinus lambertiana)	0.10	20 weeks; 6 h/d; 7 d/week	No growth reduction; no foliar injury	20
Western white pine (Pinus monticola)	0.10	20 weeks; 6 h/d;	Growth reduction in root length (5.7%)^{b}; dry weight of foliage (12.9%)^{a}, stem (9.1%)^{b}; root (12.2%)^{b}; no reduction in plant height; foliar injury	20
Sitka spruce (Picea sitchensis)	0.10	20 weeks; 6 h/d; 7 d/week	Growth reduction in root length (14.3%)^{b}; dry weights of foliage (7.1%)^{b}; stem (13.6%)^{b}; root (17.9%)^{b}; foliar injury	20
Ponderosa pine (Pinus ponderosa)	0.15	9 h/d 10, 20, 30, 60 days	4, 25, 25, and 34% (reduction) in apparent photosynthesis, respectively	17
Hybrid poplar (Populus deltoides)	0.15	6 weeks; 5 d/week; 8 h/d	50% reduction in growth, 75% foliar injury; significant reduction in dry weight of shoot, leaf, and root^{a}	10
White pine (Pinus strobus)	0.15	5 weeks continuous	Increase in total soluble sugars of primary needles; reducing sugars remained the same; decrease in ascorbic acid	1
Shortleaf pine (Pinus echinata)	0.15	5 weeks continuous	Increase in soluble sugars and ascorbic acid of the primary needles	1
Slash pine (Pinus elliotti)	0.15	5 weeks continuous	Significant increases^{a} in soluble sugars (total and reducing) and ascorbic acid of primary needles	1
Pond pine (Pinus serotina)	0.15	5 weeks continuous	No increase in total soluble sugars; slight increase in reducing sugars; and no increase in ascorbic acid of primary needles	1
Loblolly pine (Pinus taeda)	0.15	5 weeks continuous	Significant increases^{c} in soluble sugars (total and reducing of primary needles; decrease in ascorbic acid	1
Hybrid poplar (Populus maximowiczii x trichocarpa)	0.18	4 h	Adaxial leaf injury	5
Hybrid poplar (Populus maximowiczii x trichocarpa)	0.18/0.18 (O_3/PAN)	4 h	Adaxial leaf injury (injury produced by O_3 + PAN was significantly greater than the sum of that produced by O_3 and PAN exposures separately	5
Eastern white pine (Pinus strobus)	0.20	6, 12, 24, 48, 96 h (during first growing after grafting)	No direct needle injury in tolerant or sensitive clones	9
Eastern white pine (Pinus strobus)	0.20	7, 21 d	(Pre-exposed $^{14}CO_2$ fixation reduced)	21
Red pine (Pinus resinosa)	0.20	20-25 h (spring); 4-5 h (summer)	Level at which first signs of foliar injury occurred; chlorophyll decreases occurred before visual injury appeared	12
Black Hills spruce (Picea glauca)	0.25	8 h	No injury to current year needles	4
Colorado blue spruce (Picea pugens)	0.25	8 h	No injury to current year needles	4

Table 4.1 (Continued)

Species	O_3 concentration (ppm)	Exposure duration	Description of response compared to controls	Ref (see key)
Norway spruce (*Picea abies*)	0.25	8 h	No injury to current year needles	4
White spruce (*Picea glauca*)	0.25	8 h	No injury to current year needles	
Green ash (*Fraxinus pennsylvanica*)	0.25	8 h	Foliar injury	22
White oak (*Quercus alba*)	0.25	8 h	Foliar injury	22
Thornless honey locust (*Gleditisia triacanthos inermis*)	0.25	8 h	Foliar injury	22
European mountain ash (*Sorbus aucuparia*)	0.25	8 h	Foliar injury	22
American sycamore (*Platanus occidentalis*)	0.25	8 h	Foliar injury	22
Sweet gum (*Liquidambar styraciflua*)	0.25	8 h	Foliar injury	22
Pin oak (*Quercus palustris*)	0.25	8 h	Foliar injury	22
Scarlet oak (*Quercus coccinea*)	0.25	8 h	Foliar injury	22
Eastern redbud (*Cercis canadensis*)	0.25	8 h	Foliar injury	22
Eastern cottonwood (*Populus deltoides*)	0.25	110 d; 8 h/d; 5 d/week	Height growth, leaf number, and size unaffected	11
White spruce (*Picea glauca*)	0.25	110 d; 8 h/d; 5 d/week	Height growth, leaf number, and size unaffected	11
Japanese larch (*Larix leptolepis*)	0.25	110 d; 8 h/d; 5 d/week	Height growth, leaf number, and size unaffected	11
American sycamore (*Platanus occiddentalis*)	0.25	8 h	Approximately 50% reduction in apparent photosynthesis; death of palisade and upper epidermal layers; chloroplast breakdown	14
White ash (*Fraxinus americana*)	0.25	1 h	Level at which foliar injury (surface strippling) to seedlings occurred	19
European larch (*Larix dedicua*)	0.25	4, 8 h	Injury to current year needles	4
Eastern hemlock (*Tsuga cadensis*)	0.25	8 h	Injury to current year needles	4
Japanese larch (*Larix leptolepsis*)	0.25	8 h	Injury to current year needles	
Eastern white pine (*Pinus strobus*)	0.25	8 h	Injury to current year needles	4
Pitch pine (*Pinus rigida*)	0.25	8 h	Injury to current year needles	4
Arborvitae (*Thuja occidentalis*)	0.25	8 h	No injury to current year needles	4
Balsam fir (*Abies balsamea*)	0.25	8 h	No injury to current year needles	4
Douglas-fir (*Pseudotsuga menziesii*)	0.25	8 h	No injury to current year needles	4
Red pine (*Pinus resinosa*)	0.25	8 h	No injury to current year needles	4
Sugar maple (*Acer saccharum*)	0.20-0.30	2 h	Threshold level for visible foliar injury	7
Ponderosa pine (*Pinus ponderosa*)	0.25	8 h	Seedling needles exhibit maximum sensitivity (chlorotic mottle tip necrosis) during third-ninth week of age; symptoms occurred at 24 h after exposure	3
Virginia pine (*Pinus virginiana*)	0.25 0.025	2 h	Percent foliar injury to seedlings at ages 2, 4, 6, 8, and 10 weeks respectively; 55, 45, 33, 8, 6%	2
Shortleaf pine (*Pinus echinata*)	0.25 0.025	2 h	Percent foliar injury to seedlings at ages 2, 4, 6, 8, and 10 weeks respectively; 45, 10, 15, 3, 10%	2
Loblolly pine (*Pinus taeda*)	0.25 0.025	2 h	Percent foliar injury to seedlings at ages 2, 4, 6, 8, and 10 weeks respectively; 45, 25, 39, 15, 10%	2
Slash pine (*Pinus elliotti*)	0.25 0.025	2 h	Percent foliar injury to seedlings at ages 2, 4, 6, 8, and 10 weeks respectively; 35, 25, 30, 25, 0%	2

Table 4.1 (Continued)

Species	O_3 concentration (ppm)	Exposure duration	Description of response compared to controls	Ref (see key)
White birch (*Betula papyrifera*)	0.25	110 d; 8 h/d; 5 d/week	Reduction in root growth; reduced leaf number and size	11
Yellow birch (*Betula alleghaniensis*)	0.25	110 d; 8 h/d; 5 d/week	Height growth, leaf number, and size unaffected	11
Big tooth aspen (*Populus gradidenta*)	0.25	110 d; 8 h/d; 5 d/week	Height growth, leaf number, and size unaffected	11
European white birch (*Betula pendula*)	0.25	3 h	No foliar injury	22
Gray dogwood (*Cornus racemosa*)	0.25	8 h	No foliar injury	22
White dogwood (*Cornus florida*)	0.25	8 h	No foliar injury	22
Little leaf linden (*Tilia cordata*)	0.25	8 h	No foliar injury	22
Norway maple (*Acer platanoides*)	0.25	8 h	No foliar injury	22
Sugar maple (*Acer saccharum*)	0.25	8 h	No foliar injury	22
English oak (*Quercus robur*)	0.25	8 h	No foliar injury	22
Northern red oak (*Quercus rubra*)	0.25	8 h	No foliar injury	22
Shingle oak (*Quercus imbricaria*)	0.25	8 h	No foliar injury	22
Ponderosa pine (*Pinus ponderosa*)	0.25-0.35	70-90 d; 8 h/d	Apparent photosynthesis reduced with and without corresponding decrease in chlorophyll concentration	16

[a] Significant at 0.05 level of confidence.

[b] Significant at 0.10 level of confidence.

[c] Significant at 0.01 level of confidence.

[d] Insignificant at 0.10 level of confidence.

were summarized for O_3 concentration in the range of 0 to 0.25 ppm; higher concentrations were not considered relevant.

Reports are available that list tree species according to their O_3 sensitivity (Dochinger 1964, Davis and Gerhold 1976). Table 4.2 presents a ranking of tree sensitivity to ozone and is reproduced from Davis and Gerhold (1976). They discuss the usefulness of relative sensitivity lists and caution against placing great dependence on relative sensitivity rankings. The plant material (source, age, etc.), criteria for ranking (injury, growth), and the methods of study (exposure chamber, ambient air), may play an important role in the results of the experiments.

Numerous field studies have related significant changes in forest ecosystem response to ambient oxidant concentrations (Smith 1980b). The San Bernardino Mountain Study (Miller 1973, 1977) documented mortality of sensitive ponderosa (Pinus ponderosa) and Jeffrey (P. Jeffreyi) pines following bark beetle infestation of air-pollution-stressed trees. Species composition of the forest appears to be shifting in response to air pollution stress, to favor more tolerant species such as white fir. Other examples of shifts in species composition associated with oxidants or mixtures of air pollutants have been reported (Hayes and Skelly 1977, McClennahan 1978).

Unfortunately none of the above field or conceptual studies can make the necessary linkage between dose and response necessary for analyzing the regional-scale impact of an air pollutant on the productivity of an important timber species beyond their specific field situations.

Table 4.2 Relative susceptibility of trees to ozone[1]

Sensitive	Intermediate	Resistant
Ailanthus altissima	*Acer negundo*	*Abies balsamea*
Amelanchier alnifolia	*Cercis canadensis*	*Abies concolor*
Fraxinus americana	*Larix leptolepis*	*Acer grandidentatum*
Fraxinus pennsylvanica	*Libocedrus decurrens*	*Acer platanoides*
Gleditsia triacanthos	*Liquidambar styraciflua*	
Juglans regia	*Pinus attneuata*	*Betula pendula*
		Cornus florida
Larix decidua	*Pinus contorta*	*Fagus sylvatica*
Liriodendron tulipifera	*Pinus echinata*	*Ilex opaca*
		Juglans nigra
Pinus banksiana	*Pinus elliotii*	*Juniperus occidentalis*
Pinus coulteri	*Pinus lambertiana*	
Pinus jeffreyi	*Pinus rigida*	*Nyssa sylvatica*
Pinus nigra	*Pinus strobus*	*Persea americana*
Pinus ponderosa	*Pinus sylvestris*	*Picea abies*
Pinus radiata	*Pinus torreyana*	*Picea glauca*
Pinus taeda		*Picea pungens*
Pinus virginiana	*Quercus coccinea*	
	Quercus palustris	*Pinus resinosa*
Platanus occidentalis	*Quercus velutina*	*Pinus sabiniana*
Populus maximowiczii × *Trichocarpa*		*Pseudotsuga menziesii*
	Syringa vulgaris	*Pyrus communis*
Populus tremuloides		*Quercus imbricaria*
	Ulmus pavirfolia	*Quercus macrocarpa*
Quercus alba		*Quercus robur*
Quercus gambelii		*Quercus rubra*
Sorbus aucuparia		*Robinia pseudoacacia*
Syringa × *chinensis*		*Sequoia sempervirens*
		Sequoiadendron giganteum
		Thuja occidentalis
		Tilia americana
		Tilia cordata
		Tsuga canadensis

[1] Taken from Davis and Gerhold (1976).

4.3 LOSS ASSESSMENT

4.3.1 Introduction

As presented in Table 4.1, there were numerous studies in which foliar injury was the main parameter measured; few studies examined yield loss in forest trees, and even the majority of these utilized seedlings under artificial exposure chamber conditions. Hence, it is difficult to extrapolate such information to actual yield (growth expressed as height or radial increment decrease) reductions that may be experienced under forest conditions.

Several studies have been performed in which forest tree seedlings were exposed under controlled laboratory or greenhouse conditions to various doses of ozone. The collective data from these studies are presented in Tables 4.3 and 4.4.

One field study of the Blue Ridge Mountains of Virginia Forest Study by Benoit (1981) demonstrated 15 and 28% reductions in annual radial increment growth (25-year cumulative) for white pines (*Pinus strobus*) classed as intermediate and sensitive, respectively, for foliar injury over the tolerant controls. One-year-increment (1978) growth for each respective class was 15 and 45% of the tolerant trees and the 10-year increment (1968-1978) for each respective class was 21 and 40% of the tolerant controls. Due to the lack of clean air controls, these figures may be considered as conservative because they were based on tolerant tree growth in areas of known O_3 occurrence (0.05 - 0.075 ppm, 7-h av, May-September yearly).

Table 4.3 Height growth, dry weight, and foliar symptom evaluation expressed as percent of control for seedlings of 10 tree species exposed to 0, 0.05, 0.10, or 0.15 ppm O_3 for 6 h/d for 28 consecutive days. (From Kress and Skelly, in press.)

		Percent of control				
		Height growth (cm)	Dry weight (g)			
	Treatment		Top	Root	Total	Symptom
Loblolly pine	0	100a[a]	100a	100a	100a	1.0a
(*Pinus taeda* L.)	0.05	82 b	85ab	90ab	86ab	1.0a
	0.10	73 b	79 b	72 bc	78 b	1.0a
	0.15	59 c	74 b	64 b	72 b	1.3a
Loblolly pine	0	100a	100ab	100a	100ab	1.0a
(6-13x2-8)	0.05	91ab	108a	102a	107a	1.0a
(*P. taeda* L.)	0.10	91ab	95ab	93a	94ab	1.0a
	0.15	80 b	86 b	84a	86 b	1.4a
Pitch pine	0	100a	100a	100a	100a	1.0a
(*P. rigida* Mill.)	0.05	96ab	94a	84ab	92ab	1.0a
	0.10	87 b	83a	77ab	81ab	1.0a
	0.15	74 c	78a	68 b	76 b	1.4
Virginia pine	0	100a	100a	100a	100a	1.0a
(*P. virginiana* Mill.)	0.05	95a	98a	120a	102a	1.0a
	0.10	89a	97a	93a	97a	1.0a
	0.15	86a	87a	86a	87a	1.0a
Sweetgum	0	100a	100a	100a	100a	2.1 c
(*Liquidambar styraciflua* L.)	0.05	109a	91ab	88ab	90a	8.4 b
	0.10	71 b	76ab	65 bc	74ab	13.3a
	0.15	55 b	60 b	52 c	58 b	14.8a
Sycamore	0	100a	100a	100a	100a	1.3 b
(*Platanus occidentalis* L.)	0.05	104a	84a	57 b	77a	1.2 b
	0.10	73 b	43 b	27 b	39 b	4.5a
	0.15	79 b	36 b	19 b	31 b	3.8a
White ash	0	100a	100 b	100ab	100 b	1.2 b
(*Fraxinus americana* L.)	0.05	112a	125a	111a	122a	1.1 b
	0.10	91a	92 b	87 b	91 bc	4.4a
	0.15	85a	83 b	81 b	83 c	5.5a
Green ash	0	100a	100a	100a	100a	2.9 b
(*F. pennsylvanica* Marsh.)	0.05	93ab	86ab	86a	86a	4.1ab
	0.10	76 bc	71ab	75a	72a	6.7a
	0.15	70 c	64 b	75a	67a	5.8a
Willow oak	0	100a	100a	100a	100a	1.3a
(*Quercus phellos* L.)	0.05	99a	99a	94a	96a	1.0a
	0.10	96a	92a	83a	89a	2.1a
	0.15	81 b	89a	83a	87a	1.4a
Sugar maple	0	100 b	100a	100ab	100a	1.8 c
(*Acer saccharum* L.)	0.05	95 b	88a	101a	97a	1.3 c
	0.10	[illegible]	90a	116ab	93a	2.3 b
	0.15	86 c	58 b	69 b	59 b	9.6a
Yellow-poplar	0	100 b	100 b	100a	100a	3.7 b
(*Liriodendron tulipfiera* L.)	0.05	160a	147a	133a	141a	3.3 b
	0.10	108 b	111ab	96a	105a	3.9ab
	0.15	88 b	108 b	121a	110a	5.6a

[a] Values for each column/species followed by the same letter are not different at P = 0.05 according to the Duncan's Multiple Range Test.

Table 4.4 Additional studies demonstration growth effects to forest trees as induced by ozone under controlled laboratory conditions.

Species	Exposure time	O_3(ppm)	Growth effect	Reference
Sycamore (*Platanus occidentalis*)	8 h/d, 5 d/week 5 months	0.30	24% height reduction	Jensen 1973
Yellow-poplar (*Liriodendron tulipifera*)	8 h[illegible], 5 d/week 5 months	0.30	No effect	Jensen 1973
Green ash (*Fraxinus pennsylvanica*)	8 h/d, 5 d/week 5 months	0.30	36% height reduction	Jensen 1973
White ash (*F. americana*)	8 h/d, 5 d/week 5 months	0.30	7% height reduction	Jensen 1973
Sugar maple (*Acer saccharum*)	8 h/d, 5 d/week 5 months	0.30	69% height reduction	Jensen 1973
Red maple (A. rubrum)	8 h/d, 5 d/week 5 months	0.30	30% height reduction	Jensen 1973
Black walnut (*Juglans nigra*)	8 h/d, 5 d/week 5 months	0.30	12.5 height reduction	Jensen 1973
Silver maple (*A. saccharinum*)	8 h/d, 5 d/week 5 months	0.30	73% height reduction	Jensen 1973
E. black adler (*Alnus glutinosa*)	8 h/d, 5 d/week 5 months	0.30	29% height reduction	Jensen 1973
Yellow-poplar	12 h/d, 42 d	0.10	36% reduction in relative leaf area growth rate	Jensen 1981
E. cottonwood (*Populus deltoides*)	12 h/d, 42 d	0.10	53% reduction in relative leaf area growth rate	Jensen 1981
Loblolly pine (*Pinus taeda*)	6 h/d, 28 d	0.10	17% height growth reduction, 21, 13, and 20% reduction in top, root, and total dry wt.	Kress and Skelly (in press)
Loblolly pine 6-13x2-8 cross	6 h/d, 28 d	0.10	25% height growth reduction, 11, 31, and 14% reduction in top, root, and total dry wt.	Kress and Skelly (in press)
Loblolly pine 6 open poll. fam.	8 h/d, 42 d	0.05 0.10	6% reduction in height 25% reduction in height	Ward 1980 Ward 1980
	8 h/d, 21 d	0.05 0.10	2% inc. in height 28% reduction in height inc.	Ward 1980 Ward 1980
	3 h/d, 21 d	0.05 0.10	41% reduction in height inc. 50% reduction in height inc.	Ward 1980 Ward 1980
Loblolly pine	6 h/d, 28 d	0.05	5% height growth reduction (sens) 2% height growth reduction (tol)	Kress et al. 1981 Kress et al. 1981
	6 h/d, 56 d	0.05	4% height reduction (sens) 1% height growth inc. (tol)	Kress et al. 1981 Kress et al. 1981
Loblolly pine 6 families	8 h/d, 42 d	0.05 0.10	3.5% height growth reduction 12% height growth reduction	Ward 1980 Ward 1980
	3 h/d, 21 d	0.05 0.10	3.5% height growth reduction 10% height growth reduction	Ward 1980 Ward 1980
	3 h/d, 42d	0.05 0.10	11% height growth reduction 27% height growth reduction	Ward 1980 Ward 1980

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4.3.2 Appraisal of a Specific Case Study - Potential Losses to Growth of Eastern White Pine in the Western North Carolina Mountains

4.3.2.1 Introduction

Eastern white pine (*Pinus strobus* L.) is a major timber species in a 21-county area of western North Carolina (Fig. 4.2) (Cost 1975). This species is also known to be relatively sensitive to oxidant air pollution; a review of pertinent studies may be found in Section 4.2 of this report.

Eastern white pine produces 9% of the annual growth of the growing stock on commercial land and 14% of all sawtimber harvested. The value of the annual growth of eastern white pine amounts to $1,095,800 within the 21-county area.* However, considering the volumes of sawtimber as separate from the growing stock, annual sawtimber production for eastern white pine sawtimber is valued at $14,790,435.

4.3.2.2 Approach

The analysis-of-loss functions for white pine were developed exactly as described for crop species (Section 3.2 of this report), based on data from Benoit (1981) for white pine (field data for seedlings in open-top chambers). The projected change in one growth parameter (height growth) with increasing ozone concentration is summarized in Table 4.5.

Based on the growing season (June-September) average maximum 7-hour ozone concentrations for this region found in Fig. 3.3, an ozone concentration of 65

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* Eastern white pine grows throughout this mountainous terrain, and of the white pine sawtimber harvested in the entire state (Knight and McClure 1975) approximately 2.45 x 10^5 (98%) is annually harvested in this 20-county area.

Table 4.5 Change in height growth due to ozone exposure of forest tree seedlings

O_3 concentration (ppb)	% of control for height growth
	White pine[a]
20	97
40	94
60	91
80	88
100	86

[a] Equation for white pine: $Y=100 - 1.45\ X$.

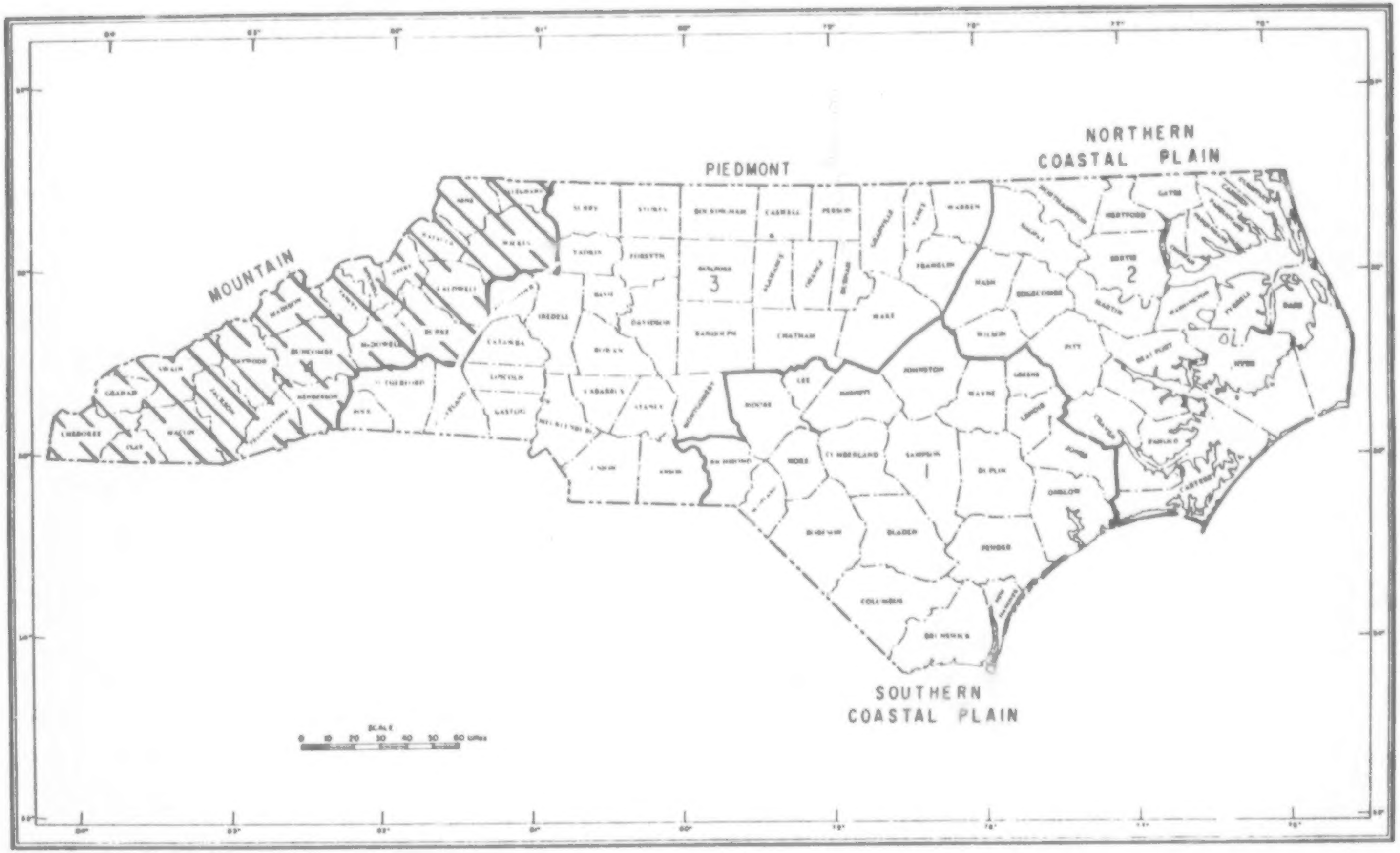
NORTHERN
COASTAL PLAIN
PIEDMONT
MOUNTAIN
SOUTHERN
COASTAL PLAIN
1
2
3
SCALE
0 10 20 30 40 50 60 Miles

Fig. 4.2 The twenty-one-county area of western North Carolina used in evaluating potential losses to eastern white pine as induced by ozone at an assumed 7-h average of 0.06 ppm for the months of April-October.

ppb was selected for this analysis. Given those conditions, white pine would be expected to experience a net reduction in height growth of approximately 10%.

4.3.3 Uncertainty Associated with Interpretation and Extrapolation of Effects Data to Forests and Forest Ecosystem Loss Assessments

- Most studies that demonstrated growth reductions in forest trees were done with seedlings exposed under controlled fumigation conditions; effects on mature forest trees under forested conditions may be considerably different.

- The relationship between foliar injury and subsequent yield reductions are not well understood; consequently foliar injury as a tool for detection of loss under forest conditions is of limited value.

- Genetic diversity in forest species is considerable, and test data on limited numbers of seedlings per species in various trials of limited value in assessing losses in mature forest stands.

- Interactions of environmental factors which are known to alter sensitivity are difficult to assess for individual forest trees, competing species within a stand, stands, communities, and forest types. The diversity of forests (species, terrain, soil, water tables, etc.) causes difficulty in evaluating "forest" effects.

- Forest type varies from region to region (and in many areas varies on a small spatial scale), and losses are assumed to occur uniformly by ozone isopleth on a county-area basis; likewise, ozone isopleths are considered to be uniform on at least a county-area basis.

- Ambient doses vary considerably over a temporal and spatial dispersion pattern.

- Economic appraisals vary with fluctuation in national timber-pricing trends and more importantly with local harvesting practices and stumpage value; losses in one area may not be of management importance in another region of the country.

- The data (as presented) pertain to O_3-induced effects alone; in areas where SO_2, NO_2, or hydrogen fluoride (HF) are present, increased damage may be expected to occur.

o Long-term low dose exposures throughout the entire year as well as through sucessive years may induce reduced tree vigor, which has not been considered in this report (only April through October, 7-hour daily averages). Effects from year to year may be cumulative and lead to attack of weakened trees by biotic or abiotic stress-inducing agents.

4.4 INTEGRATION OF EFFECTS OF MULTIPLE POLLUTANTS ON FORESTS

Because of the large uncertainties associated with estimates on forest growth impacts caused by ozone exposure and the current inability to estimate direct effects of acidic deposition on forest growth, a discussion of the combined potential long-term effects of acidic deposition and gaseous pollutants (such as ozone) on forest productivity must, of necessity, be only qualitative at the present time.

Figure 4.3 summarizes the various mechanisms by which such a combination of pollutants might affect forest productivity. This figure indicates that the net effect on forest growth may be expected to represent the sum of a series of positive and negative inputs to the system. The net effect on productivity will depend strongly on those site-specific factors controlling forest response to any individual mechanism, assuming no interaction between mechanisms. By examining the mass of evidence suggesting (1) potential cation nutrient depletion, (2) potential ozone impact on sensitive forest species, and (3) potential benefits due to sulfur and nitrogen inputs, certain patterns are suggested.

The negative impacts appear to reach their greatest potential in commercial forests of the southeastern coastal plain, the Mississippi River valley, the Appalachian mountain chain, and the upper Ohio River valley. In each of these regions, nitrogen inputs to forests would be expected to partially offset negative impacts, while sulfur inputs currently exceed forest

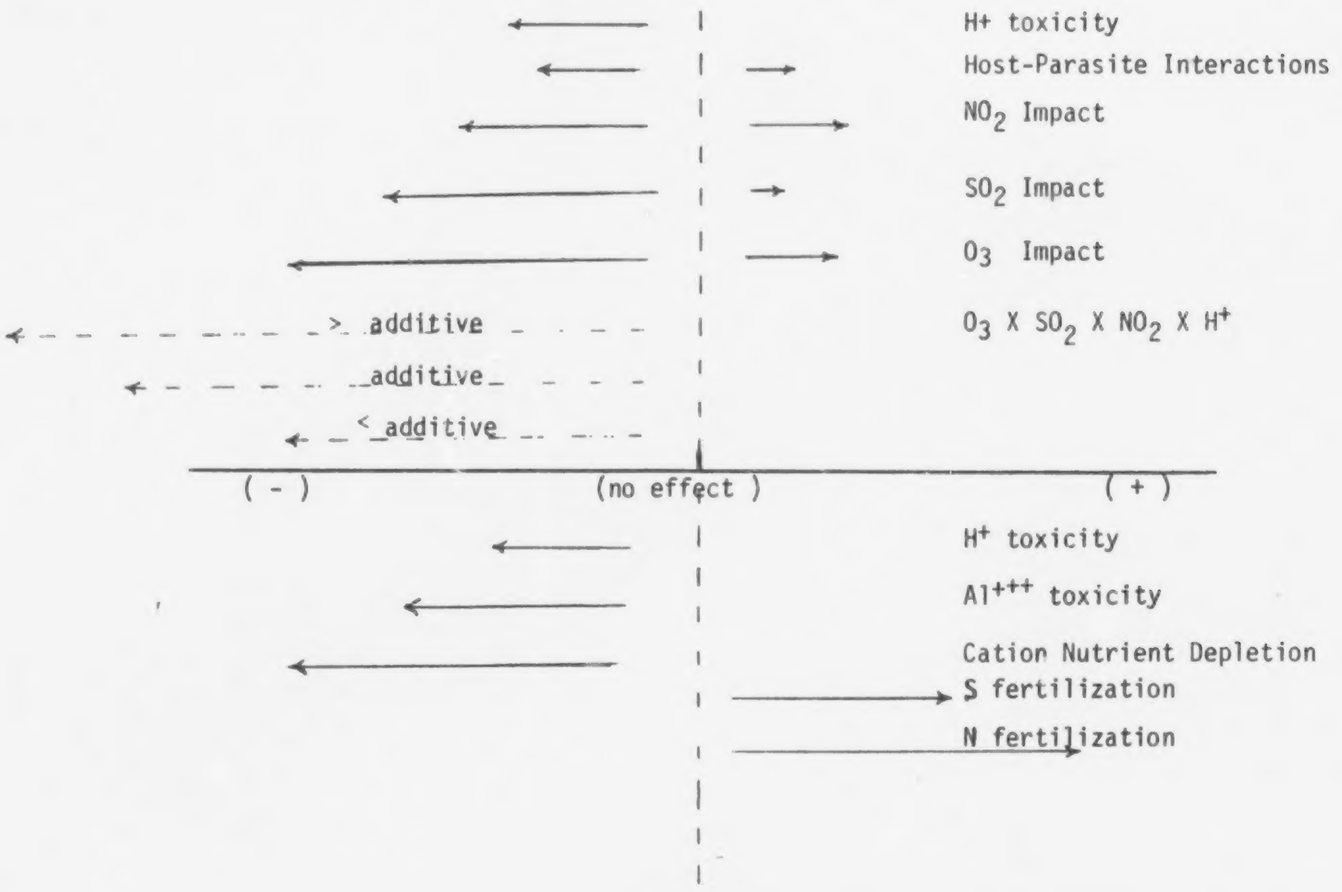
H+ toxicity
Host-Parasite Interactions
NO_2 Impact
SO_2 Impact
O_3 Impact
> additive
additive
< additive
O_3 X SO_2 X NO_2 X H^+
(-)
(no effect)
(+)
H^+ toxicity
Al^{+++} toxicity
Cation Nutrient Depletion
S fertilization
N fertilization

Fig. 4.3. A schematic representation of the mechanisms interacting to impact forest growth as a result of the combined inputs of wet and dry deposition of atmospheric pollutants.

growth requirements and probably do not represent a significant influence either positive or negative at this time.

As pointed out above, such an analysis assumes no interaction between the various factors influencing forest productivity, either positively or negatively. This assumption is probably not valid in the case of SO_2, ozone, and oxides of nitrogen, where the initial data available on multiple pollutant exposure (all based on crop species) suggest that greater or less than additive effects may more commonly result from multiple pollutant exposures than do additive responses (Reinert et al. 1981, Heggestad et al. 1981). Complicating the issue even further is the fact that such interactions depend on the species and the ratios of sulfur dioxide, ozone, and nitrogen dioxide in the mixture (Reinert et al. 1981). Only three studies are reported in the literature dealing with the potential interaction of any of these gaseous pollutants with acid rain [Shriner 1978 - (O_3); Irving and Miller 1981 - (SO_2); and Jacobson et al. 1980 and Troiano et al. 1981 - (O_3)]. In all three cases, agricultural species were studied. In the two studies involving acid rain and ozone, nonadditivity was observed, while the study involving acid rain and SO_2 was essentially additive in effect.

Recent studies using tree rings to analyze growth (Johnson, et al. 1981: Puckett, in press) showed significant declines in the growth of several tree species in various parts of the northeastern United States. These trends could not be accounted for by climatic patterns alone, leading the authors to suggest that acid rain may have been a factor. It would appear to be a viable hypothesis, given patterns of deposition discussed elsewhere in this report, that such growth trends represent the net result of the combination of air quality factors (ozone, SO_2, NO_2, $H+$, $SO^{4=}$, NO_3^-, to which tree species are exposed.

In summarizing the potential long-term effects on forest productivity of both acid deposition (wet) and gaseous (dry) deposition, it is possible at the present time to state only that such interactions may potentially occur and that their probability of occurrence is greatest in those regions of the eastern United States outlined above. Further qualification and eventual quantification of these potential effects must await further study of the mechanisms involved and of the relative importance of those mechanisms to forest growth response.

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Appendix GG

REGIONAL ASSESSMENT OF AQUATIC RESOURCES AT RISK FROM ACIDIC DEPOSITION

I. EXECUTIVE SUMMARY

This study of the relationship between aquatic resources and recent as well as projected trends in acidic deposition has been carried out as part of the Office of Technology Assessment's investigation of the significance of long-range transport air pollution. Our objective was to develop a base-line inventory of acid-sensitive aquatic resources in the eastern United States; to evaluate recent trends in key measures of aquatic environments; and, to the extent possible, estimate the potential magnitude of lake and stream alterations (deterioration or recovery) associated with three alternative scenarios of future acidic deposition.

The approach used to develop the inventory of sensitive lakes and streams was as follows: (1) Soil and watershed criteria were used throughout 27 eastern States to develop a list of individual counties sensitive to acidic deposition; (2) these areas were aggregated into 14 acid-sensitive regions with representation in 23 of the 27 states; (3) the total number of lakes greater than 15 acres in size, and total miles of first- and second-order streams, were estimated by a systematic inventory (on a 12 to 18 percent sub-sample) of each region using USGS maps; and (4) the number of low-alkalinity lakes and streams was estimated from a series of regional water quality surveys summarized as percentages of the surface water resource falling into each of five widely-recognized alkalinity classes.

The results show an estimated 17,000 lakes in the fourteen defined sensitive regions in the eastern U.S. Of these, approximately 9,400 (± 20%) are estimated to have alkalinity levels of less than 200 microequivalents per liter, a level indicating either sensitivity to acid inputs or existing alteration by these inputs. For streams, a total 117,000 miles of first- and second-order streams occur in the fourteen regions, of which 60,000 miles (± 20%) can be categorized as aquatic resources either already altered or seriously at risk.

Alkalinity data from unimpacted areas of the Canadian Shield -- areas which are geologically comparable to the heavily impacted areas of the U.S. -- and historical time-trend data for alkalinity in the affected areas, indicate that a major portion of the lakes currently in the lowest alkalinity classes (from negative values up to 40 microequivalents per liter) are already "acid-altered" rather than simply currently "sensitive". An estimated 3,000 lakes and 23,000 miles of stream (± 20%) meet this criterion in the eastern U.S.

The several approaches available for describing recent acidification rates are reviewed briefly, but none is found to be appropriate for estimating benefits from future abatement of sulfate deposition. The watershed processes which mediate acid inputs are also reviewed, and are found to be theoretically sufficient but lacking quantitative solutions at this time. As an alternative, a preliminary model describing alkalinity trends was developed using the theory available and a simple "alkalinity impact parameter" (AIP) quantified by both theoretical and field procedures. Although not yet fully tested, it suggests that lakes and stream alkalinities are likely to change with future changes in sulfate deposition.

This simple model was used to estimate the effects on aquatic resources of 3 possible future acid deposition scenarios. Scenario 1, a 10% increase in sulfate deposition by 2000, is estimated to result in a about a 5 to 15 percent increase in the number of lakes and streams included in the "acid-altered" category. Under the second scenario, a 20% decrease in sulfate deposition by the year 2000, 10 to 25 percent of the lakes and streams are estimated to experience some recovery from the "acid-altered" category. Scenario 3, a 35% decrease in sulfate deposition by 2000, is estimated to result in some recovery for 15 to 40 percent of aquatic resources.

Other factors that cannot yet be considered fully are known to interact with alkalinity trends during recovery or degradation, and further research is needed on these questions. The results presented here should be viewed as probable within a numerical confidence interval of ± 50%.

II. AN INVENTORY OF LAKES AND STREAMS IN AREAS SENSITIVE TO ACID DEPOSITION

1. The Sensitive Regions of Interest

Studies by Hendrey et al. (1980), by McFee (1980) by Cowell et al. (1981), and by Olson et al. (1981), have helped to define those regions with soil neutralizing and buffering characteristics which render them "sensitive" to atmospheric acidic inputs. The concept of sensitivity is used here to refer to the likelihood of a response by the resource (soils or water) to continued acid input.

These studies show that glaciated regions having soils with very low cation exchange capacity (usually low in such base minerals as calcium and magnesium also tend to have relatively low surface-water pH where atmospheric acid inputs are high. These low-base soils prevail throughout the northeast, from Minnesota to Maine. For the non-glaciated regions (i.e., areas south of Pennsylvania), the low levels of neutralizing bases in the substrate bedrock are strongly related to the low alkalinity in stream waters, and associated low pH (Hendrey et al. (1980).

The more recent work by Johnson (1980) and D.W. Johnson, et al. (1981) indicates, however, that soils must also be low in sulfate retention for a flushing of sulfate (and associated acidity) to take place. Thus, earlier maps of regional sensitivity based on low cation exchange capacity must be re-interpreted using the additional criterion of soil sulfate retention. The regions of interest for evaluating surface water resources at risk must, therefore, be defined by aggregating all three types of field data -- bedrock geology, surface soil cation exchange capacity, and overall sulfate retention. Appropriate procedures have been developed and described by another contractor in the OTA study.[1]

Finally, it should be noted that the scope of study for aquatic resources at risk is being limited to the United States east of the Mississippi River. Surface waters in other parts of the U.S. may be experiencing effects from acid deposition (McColl 1980, Lewis and Grant 1980), but information about the geographic extent of the sensitive regions in California and Colorado is not available. At this time, no assessment methodology can be applied to surveys of water quality characteristics in these regions.

2. Establishing Homogeneous Regions for Sampling

Since, within the financial resources of OTA, it is not possible (and probably not necessary) to census the surface water resources of the entire eastern U.S., fourteen reasonably homogeneous regions have been identified for purposes of subsampling and inventory.[2] See Figure 1. For these reasons we counted and measured (length for streams, and number/areas for lakes) enough of the resources of each region to obtain a statistically representative estimate of its aquatic resources.[3]

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[1] An additional criterion has been introduced in this study for mountainous regions: the presence of strong topography, and elevations exceeding 2000 feet on 10 percent or more of the county, can function to override the "high sulfate retention" criterion. This change adds a series of counties in Virginia and North Carolina not listed by Olson et al. (1981).

3. Inventory Results by Homogeneous Sampling Area and by State

Estimates of aquatic resources in each of the fourteen sensitive regions (lakes, and first- and second-order streams) are shown in Table 1. The greater part of the ponded waters are in the Adirondacks, New England and Maine (Regions 1 to 4), and the northern glaciated areas of the Great Lakes States (Regions 11 to 14).

The overall result for lakes is that just over 17,000 ponded water-bodies exceeding 15 acres (within a confidence level of 10 to 20%) occur in the defined sensitive regions.

Flowing water resources of stream orders 1 and 2 are more evenly distributed than lakes, with rather similar densities in both the glaciated states in the north and in the unglaciated regions of the South. The total of 117,423 miles of first- and second-order streams appears to be reproducible within plus or minus 10%.

An alternative summary format, showing the area of each State included in one or more of the fourteen sensitive regions, is presented in Table 2. The proportion of each state with resources potentially at risk ranges from 0 to

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[2] These homogeneous regions are derived in large part from well-established biological/physiographic regions for the eastern United States.

[3] Subsampling of small subregions, where up to 100% of the area could be sampled, indicates that a 10% areal subsample of streams meets the objective of reproducibility within plus or minus 10%, and a 15 to 20% subsample of lakes also meets this objective, depending on local geology.

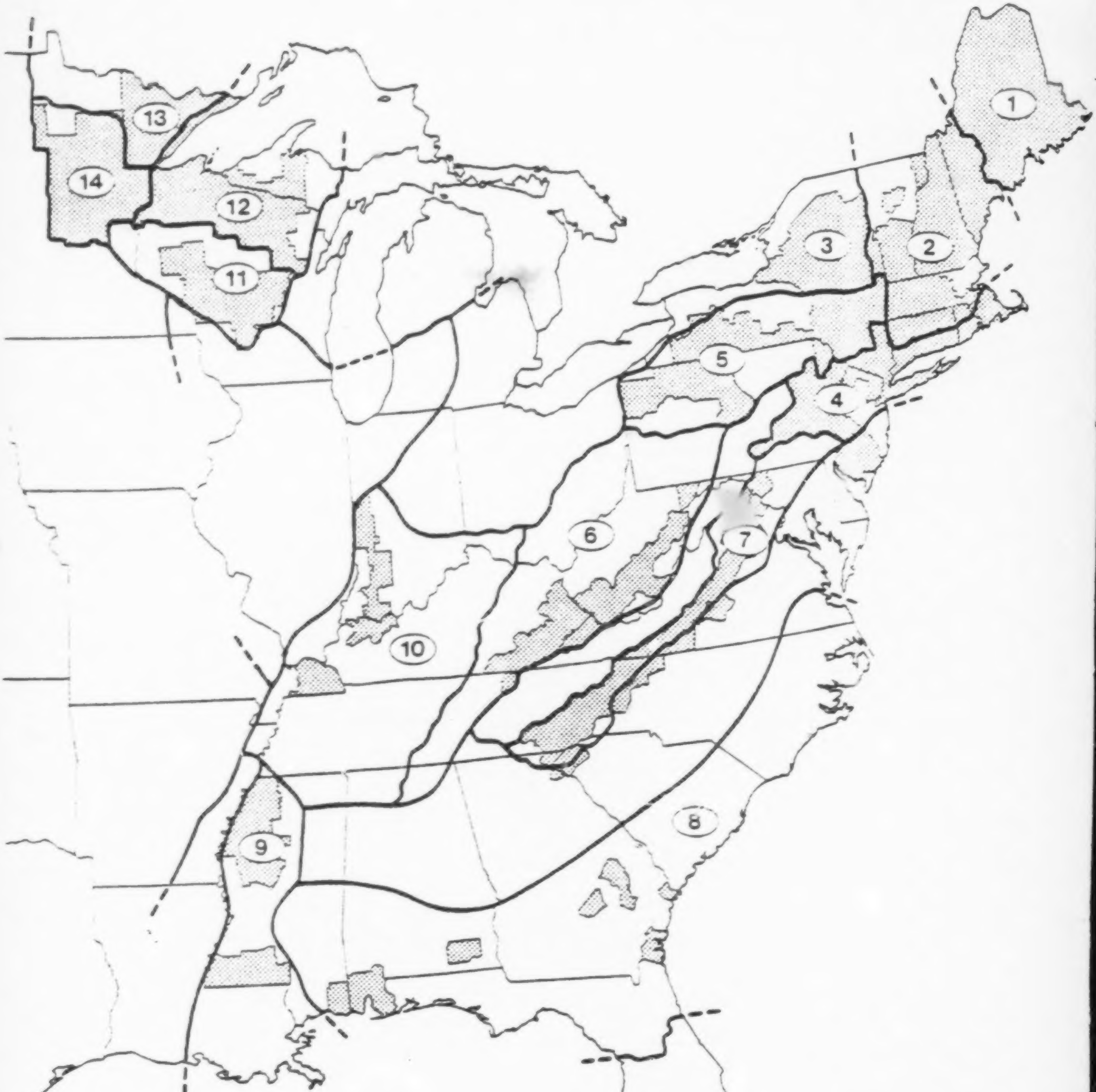

Figure 1. Homogeneous sampling areas as modified in this report from Braun (1950) and Fenneman (1938), and areas identified as sensitive to acid rain (stippled). Regional numbers correspond to names shown in Table 10.

Table 1. Total lake and stream resources estimated for the acid sensitive regions of the Eastern U.S. (see Figure 9). Lake estimates have a reproducibility of 10 to 20%, and stream estimates approximately 10%.

Region	Sensitive Area (mi²)	- -Total Lakes- -		-Total Streams (mi)-	
		Number	Acres	1st Order	2nd Order
1. Eastern Maine	26,398	1,425	582,825	9,714	3,485
2. Western New England	29,666	1,543	763,785	15,308	4,569
3. Adirondacks	14,066	1,139	231,217	5,289	3,024
4. East Pennsylvania S. New England	20,947	1,320	118,800	8,400	2,556
5. W. New York/Pennsylvania	25,051	376	16,920	7,114	1,678
6. Appalachian Plateau	16,190	13	29,510	7,350	2,299
7. Blue Ridge/ Great Smoky Mts.	20,964	126	14,868	10,901	3,396
8. Coastal Plain	9,264	241	8,917	1,547	713
9. Lower Mississippi	13,075	170	56,610	5,374	1,255
10. Indiana/Kentucky	8,603	9	603	2,805	989
11. Central Wisconsin	12,141	583	187,726	2,683	728
12. Wisconsin/Michigan Highlands	19,229	5,307	801,357	5,037	1,737
13. Northeast Minnesota	10,560	1,637	473,093	1,637	475
14. Central Minnesota	18,870	3,170	323,340	4,831	2,529
TOTALS	245,024	17,059	3,609,571	87,990	29,433

Table 2. State areas (in square miles), shown along with number and area of counties in each state identified as meeting aquatic resource sensitivity criteria (see Table 3), and the proportion of the states meeting these criteria.

State	Total area (mi^2)	Number sensitive counties	Sensitive area (mi^2)	% of total state area
States with some sensitive areas				
1. Alabama	51,609*	4	4,184*	8
2. Connecticut	5,009	all(8)	5,009	100
3. Georgia	58,876	10	4,393	8
4. Indiana	36,291	11	4,339	12
5. Kentucky	40,395	29	10,692	27
6. Maine	33,215	all (16)	33,215	100
7. Maryland	10,577	2	1,338	13
8. Massachusetts	8,257	11	6,796	82
9. Michigan	58,216	3	3,290	6
10. Minnesota	84,068	19	29,430	35
11. Mississippi	47,716	25	14,317	30
12. New Hampshire	9,304	all (10)	9,304	100
13. New Jersey	7,836	12	4,173	53
14. New York	49,576	39	33,938	69
15. North Carolina	52,586	20	8,748	17
16. Pennsylvania	45,333	26	17,719	39
17. Rhode Island	1,214	all (5)	1,214	100
18. South Carolina	31,055	2	1,172	4
19. Tennessee	42,244	1	468	1
20. Vermont	9,609	6	4,760	50
21. Virginia	40,817	25	10,860	27
22. West Virginia	24,181	13	7,585	31
23. Wisconsin	56,154	31.5	28,080	50
States with no identified sensitive areas				
1. Delaware	2,057	0	0	0
2. Florida	58,560	0	0	0
3. Illinois	56,400	0	0	0
4. Ohio	41,222	0	0	0
TOTAL AREA	962,377	328.5	245,024	25

* Areas of states and counties taken from Computer Graphics, Geography Division, Oak Ridge National Laboratory.

100 percent, averaging 25 percent for the 27 states in the region.

III. REGIONAL pH AND ALKALINITY OF SURFACE WATERS

To assess aquatic resources either at risk (i.e., highly sensitive) from or already altered by acid deposition, quantitative water chemistry measures are needed to characterize lakes and streams in the regions of interest. A broad spectrum of watershed types (which control the buffering capacity and susceptibility of the drainage waters) can be found in each of the defined sensitive regions. For example, some lakes within the Adirondacks show no effects of acidification (Wright _et al_. 1980), although the terrain is generally sensitive and deposition of acidic substances is high. Given the general inability (at this time) to predict responsiveness to acid inputs based on watershed characteristics alone, the alternative of using empirically observed distributions of lake and stream data has been applied in order to integrate local deposition and acid-neutralizing processes of watersheds.

In this report, lake and stream water quality has been evaluated using the most recent "alkalinity" measurements available from the sampling areas. Alkalinity is the capacity of water to neutralize acid, as measured by titrating a sample of the water with a known strong acid (see Lerman 1978, for additional detail).

Reports by Kramer (1976), Harvey _et al_. (1981), and others have divided the spectrum of their sensitivity measure into a series of intervals referred to as sensitivity classes or categories. Each approach differed slightly from the others, leading the authors of the U.S.-Canada Aquatic Impacts Assessment Subgroup (U.S.-Canada 1981) to propose the following classification for lakes:

Class	Alkalinity (ueq/l)
I Acidified	≤ 0
II Extreme Sensitivity	0 - 39
III Moderate Sensitivity	40 - 199
IV Low Sensitivity	200 - 499
V Not Sensitive	≥ 500

A broad consensus among authors using various sensitivity classifications indicates that 200 ueq/l can be treated as a borderline between aquatic systems to be considered sensitive, and relatively insensitive, to acid precipitation. There is a significant probability that lakes and streams below this level will be affected by acid precipitation; those above this level are unlikely to be affected in the near future. On this basis, 80% of the waters in the Adirondacks are either affected now or are "moderately sensitive". The low alkalinities of the northeastern waters (which are all in geologically sensitive areas) probably reflect the greater duration and intensity of acid deposition in these regions. Regions to the west in Wisconsin and Minnesota show lower frequencies of low-alkalinity waters, apparently reflecting the relatively unaltered conditions. Here, 13 to 48% of the waters can be regarded as at least moderately sensitive and could be affected by future atmospheric acid inputs.

The alkalinity distribution of lake and stream data from 10 different areas of the eastern U.S. are listed in Table 3. These represent current conditions -- that is, alkalinity observed under current levels of acid deposition.

A historic background distribution of surface water alkalinities and pH can be used to interpret the results presented in Table 3. The report by the U.S./Canada Subgroup on Aquatic Impacts (Glass et al. 1981) presents data on three regions in northwestern Ontario and one in Minnesota in which acidic deposition is low and of relatively recent duration (see also Glass and

TABLE 3. Percentage of lakes and streams in various sensitivity classes, classified for six locations in the eastern U.S. and four in the upper Midwest.

Region	Percentage of Lakes and Streams Sampled					Total Number Lakes/Streams Sampled
	Acidified (≦ 0 µeq/ℓ)	Extreme Sensitivity (0-39 µeq/ℓ)	Moderate Sensitivity (40-199 µeq/ℓ)	Low Sensitivity (200-499 µeq/ℓ)	Not Sensitive (≧ 500 µeq/ℓ)	
Lakes						
Adirondack[1]	23	17	40	15	5	432
Northern Wisconsin[2]	3	20	19	32	26	276
Northern Minnesota[3]			48	35	17	85
Streams						
N.W. Pennsylvania[4]	11	11	39	28	11	18
E. Pennsylvania[4]		19	19	31	31	26
Connecticut[5]	4	4	48	44		23
Maine[5]		6	75	19		16
NC/VA[6]		5	59	33	3	38
Jackson Co. (Wis.)[7]		8	35	18	39	80
Oneida Co. (Wis.)[8]		2	11	33	54	195

[1] Pfeiffer and Festa (1980). (1979 data only.)
[2] Eilers et al. (1979).
[3] Glass and Loucks (1980).
[4] Arnold (1981). (Acid mine drainage effects were avoided.)
[5] Haines (1981).
[6] Hendrey et al. (1980). (See text for sampling base.)
[7] Klick and Threinen (1968).
[8] Andrews and Threinen (1966). (Includes only streams with watershed areas ≦ 3 mi².)

Loucks, 1980). These data are summarized and combined in Table 4 to obtain an indication of the distribution of lakes among alkalinity classes (as shown in the bottom line).

Differences between streams and lakes in observed frequencies of low-alkalinity waters for the same soil types, bedrock, and acid loadings, are important to this study. The northern Wisconsin lakes and the streams in two northern Wisconsin counties are most comparable -- 23% of the lakes are lower than 40 ueq/l, while for the streams, a value near 100 ueq/l would cut off a similar 23% of the observations. Thus, the indicated difference between these lakes and streams is 60 ueq/l. N.W. Pennsylvania streams and Adirondack lakes also are substantially comparable. Forty percent of the lakes in the Adirondacks are below 40 ueq/l; a value of 100 ueq/l or more is required to cut off a similar percentage of streams in N.W. Pennsylvania. Again, the difference between lakes and streams in this low-alkalinity range appears to be in the order of 60 ueq/l, but one should attach a confidence limitation of at least ± 20% to this estimate.

IV. ESTIMATES OF SURFACE WATERS AT RISK OR ALREADY ALTERED

The previous two sections have presented (1) an estimate of the total lake and small-stream inventory for the 14 sensitive regions, and (2) the distribution of total alkalinities -- a measure of surface water sensitivity to acid inputs -- among the lakes and small streams of a subset of these regions. This section will present and evaluate the specific inventory of sensitive or acid-altered lakes and streams in the 14 sensitive regions.

The total lake and stream resources in each alkalinity class can be obtained by cross-multiplying the regional inventory of lakes and streams by regional alkalinity distributions. These are shown in Tables 5 and 6. The

Table 4. Number and percentage of lakes from low pollutant regions of the Canadian Shield, by alkalinity classes.

	NUMBER OF LAKES IN EACH ALKALINITY CLASS					
County or District	Acidified (≤0 µeq/ℓ)	Extreme Sensitivity (0-40 µeq/ℓ)	Moderate Sensitivity (40-199 µeq/ℓ)	Low Sensitivity (200-499 µeq/ℓ)	Not Sensitive (≥500 µeq/ℓ)	Total Number of Lakes
Kenora District			12	22	54	88
Northern Minnesota			41	30	14	85
Rainy River District		3	63	19	14	99
Thunder Bay District	1	3	34	37	61	136
TOTAL NUMBER	1	6	150	108	143	408
% OF TOTAL	.3	1.4	36.9	26.3	35.1	

Table 5. Total lake resources present and at risk in the acid sensitive regions of the Eastern United States.

Region	Sensitive Area (mi²)	Total Number Lakes (±10%)	Percentage of Lakes <200 ueq/ℓ (± 10-20%)	Calculated No. Lakes at Risk (±20%)
1. Eastern Maine	26,398	1,425	(80)[1]	1,140
2. Western New England	29,666	1,543	(80)[1]	1,234
3. Adirondacks	14,066	1,139	80	911
4. East Pennsylvania S. New England	20,947	1,320	(80)[1]	1,056
5. W. New York/Pennsylvania	25,051	376	(80)[1]	301
6. Appalachian Plateau	16,190	13	--[4]	--
7. Blue Ridge/ Great Smoky Mts.	20,964	126	--	--
8. Coastal Plain	9,264	241	--	--
9. Lower Mississippi	13,075	170	--	--
10. Indiana/Kentucky	8,603	9	--	--
11. Central Wisconsin	12,141	583	(42)[2]	245
12. Wisconsin/Michigan Highlands	19,229	5,307	42	2,228
13. Northeast Minnesota	10,560	1,637	48	786
14. Central Minnesota	18,870	3,170	(48)[3]	1,522
TOTALS	245,024	17,059		9,423

[1] In the absence of other data, alkalinities from the Adirondacks have been used.

[2] In the absence of other data, alkalinities from N. Wisconsin have been used.

[3] In the absence of other data, alkalinities from N. Minnesota have been used.

[4] No estimate is being made for regions with fewer than 250 lakes.

Table 6. Total first- and second-order stream resources present and at risk in the acid-sensitive regions of the Eastern United States. The confidence interval which would include 95% of replicated estimates is estimated to be ± 20%.

Region	Sensitive Area (mi^2)	Total Streams (mi) 1st Order	Total Streams (mi) 2nd Order	Percentage of Stream (± 10%) <200 ueq/l	Calculated Total Miles (± 20%) of 1° Streams at Risk	Calculated Total Miles (± 20%) of 2° Streams at Risk
1. Eastern Maine	26,398	9,714	3,485	81	7,868	2,823
2. Western New Eng.	29,666	15,308	4,569	56	8,573	2,559
3. Adirondacks	14,066	5,289	3,024	(56)[1]	2,962	1,693
4. East Penna. S. New Eng.	20,947	8,400	2,556	38	3,192	971
5. W. New York/Penna.	25,051	7,114	1,678	61	4,340	1,024
6. Appalachian Plat.	16,190	7,350	2,299	(61)[2]	4,484	1,402
7. Blue Ridge/ Smoky Mts.	20,964	10,901	3,396	64	6,977	2,173
8. Coastal Plain	9,264	1,547	713	(43)[3]	665	307
9. Lower Miss.	13,075	5,374	1,255	(43)[3]	2,311	540
10. Indiana/Kentucky	8,603	2,805	989	(43)[3]	1,206	425
11. Central Wis.	12,141	2,683	728	43	1,154	313
12. Wisconsin/Mich. Highlands	19,229	5,037	1,737	13	655	226
13. Northeast Minn.	10,560	1,637	475	(13)[4]	213	62
14. Central Minn.	18,870	4,831	2,529	(13)[4]	628	329
TOTALS	245,024	87,990	29,433		45,228	14,847

1. In the absence of other data, alkalinities from W. New England and W. New York/Pennsylvania have been used.

2. In the absence of other data, alkalinities from W. New York/Pennsylvania have been used.

3. In the absence of other data, alkalinities from Central Wisconsin have been used.

4. In the absence of other data, alkalinities from Northern Wisconsin have been used.

lake inventory estimates for Regions 6 to 10 in the southern U.S. have been omitted because of the small number involved and the incompleteness of regional alkalinity data. In regions where a large number of lakes and streams are present, but for which no alkalinity data exist, alkalinity data from the closest region with comparable soils and geology have been used. The alkalinity distributions used for the various regions, representing in some cases a composite of adjacent regions, are shown in Table 7 for both lakes and streams.

The geographical distribution of the surface water resources, and of the categories of sensitive resources, is shown by the maps in Figures 2 and 3. The size of the circles indicates magnitude of the lake and stream resource in each region sampled; the size of the wedge indicates the proportion of the resource in each of the sensitivity classes, numbering clockwise (I, II, III, IV and V) from acidified to not sensitive.

Approximately 55% of the lakes and 50% of the streams in the 14 regions are either already acidified, or extremely or moderately sensitive to acid deposition.

Although many of these low-alkalinity lakes can be viewed as being simply "at risk of acidification" from continued or increased acidic deposition, present knowledge about the alteration of soil and water resources indicates that many of the low alkalinities now being observed represent an already "altered state", the consequence of past years of acidic deposition (Patrick *et al*. 1981). Indeed, many such water bodies, streams as well as lakes, probably should be considered "in equilibrium" with recent acidic inputs, and, therefore, probably beyond the stage at which they can be considered "at risk" in the usual sense of this phrase.

Thus, for an inventory of low-alkalinity surface waters, it is also of

Table 7. Regional composite estimates of the percentage of lakes and streams in various alkalinity classes, based on the data presented in Tables 12 and 13.

	Sensitivity Class (alkalinity expressed in μeq/ℓ)				
Regional Groups	Acidified ≦ 0	Extreme 0 - 30	Moderate 40 - 199	Low 200 - 499	Not ≧ 500
LAKES					
I. Adirondacks W. New York/Penn.	23	17	40	15	5
II. Maine W. New England E. Penn./S. New England	13	19	30	24	15
III. Wisconsin/Michigan Highlands Central Wisconsin	3	20	19	32	26
IV. Central Minnesota Northeast Minnesota	1	2	36	26	35
STREAMS					
I. Adirondacks W. New York/Penn.	11	11	39	28	11
II. Maine W. New England E. Penn./S. New England Appalachian Plateau Blue Ridge/Great Smokies	1	8	51	31	9
III. Coastal Plain Indiana/Kentucky Lower Mississippi Central Wisconsin	0	8	35	18	39
IV. Wisconsin/Michigan Highlands Central Minnesota Northeast Minnesota	0	2	11	33	54

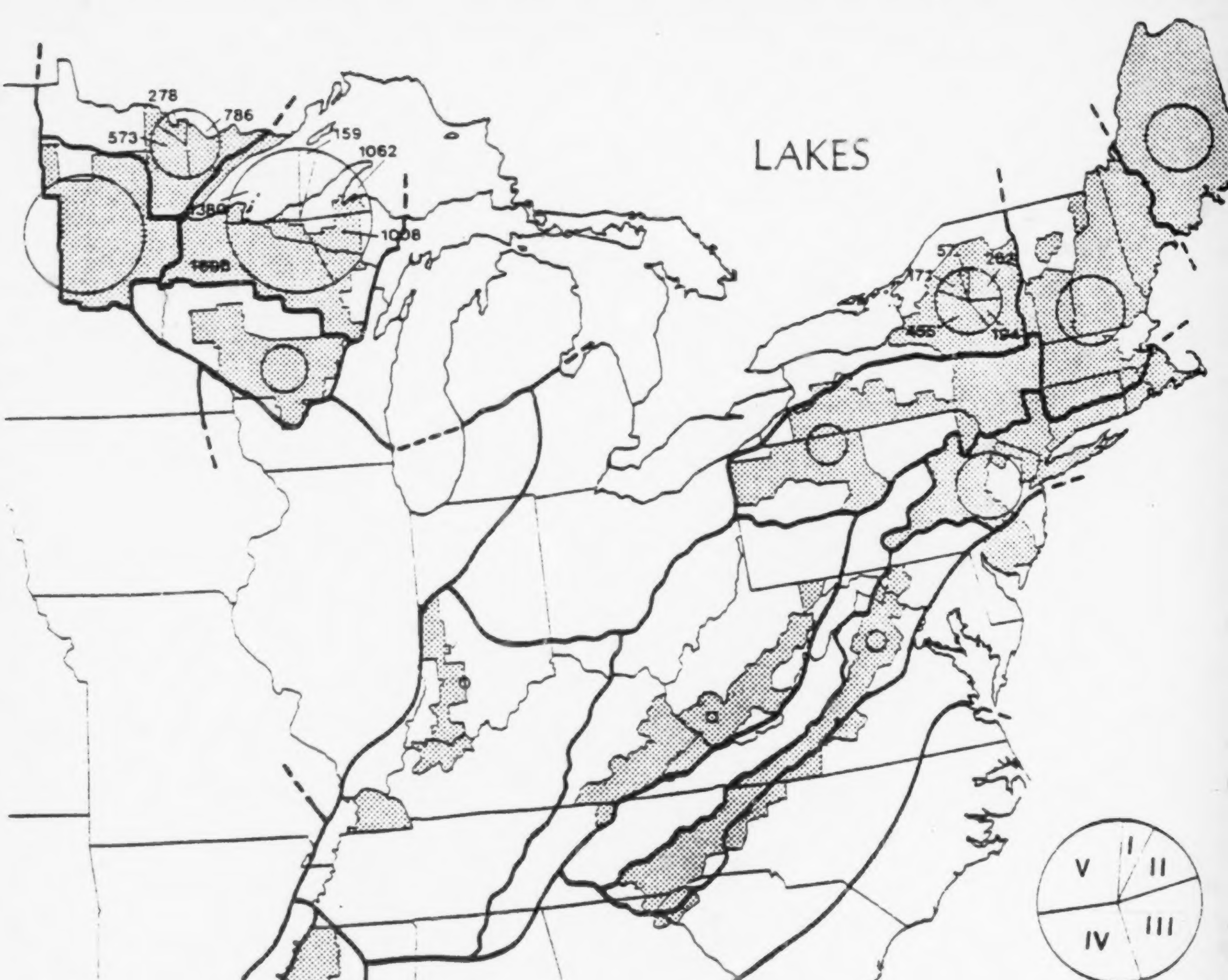
LAKES
278
786
573
159
1062
1380
1008
1698
57
263
173
455
194
V
I
II
III
IV
I - Alk. <0 µeq/ℓ
II - Alk. 0-39 µeq/ℓ
III - Alk. 40-200 µeq/
IV - Alk. 200-500 µeq
V - Alk. >500 µeq/ℓ

Figure 2: Map of the eastern United States depicting the number of lakes located in the acid-sensitive areas of the homogeneous regions. For these regions the number of lakes have been sub-divided into sensitivity classes based on alkalinity distribution. Sensitivity classes are designated as follows: I-acidified; II-extreme; III-moderate; IV-low; V-not sensitive. (Circle size shows the magnitude of lake resource. Circles not sub-divided indicate local akalinity measurements were not available for the regions to categorize the lakes among the four sensitivity classes.)

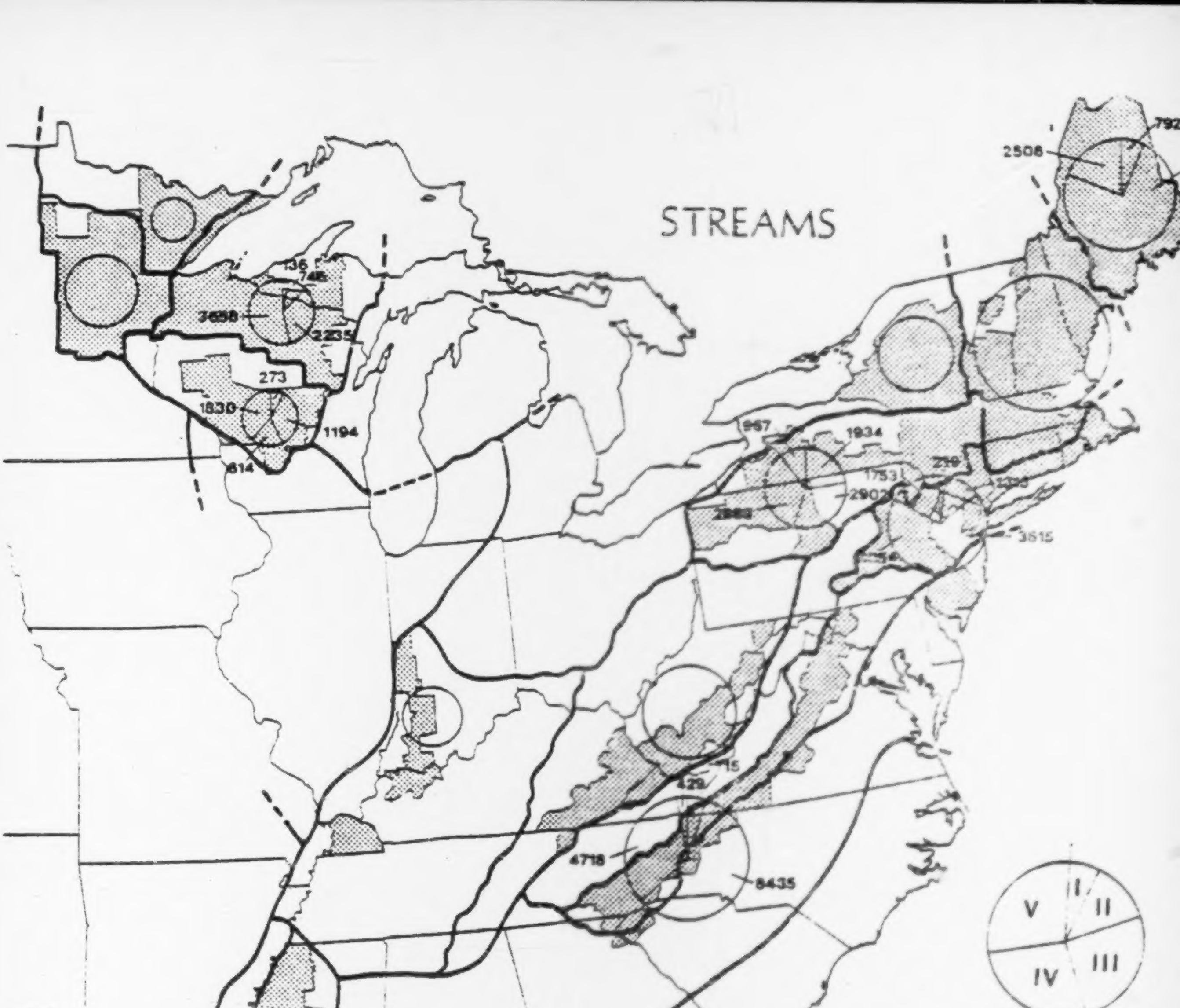
STREAMS
792
2508
136
746
3658
2235
273
1830
1194
814
967
1934
1753
2902
219
2883
3615
15
429
4718
8435
V
I
II
III
IV
I - Alk. <0 µeq/ℓ
II - Alk. 0-39 µeq/ℓ
III - Alk. 40-200 µeq/ℓ
IV - Alk. 200-500 µeq/ℓ
V - Alk. >500 µeq/ℓ

Figure 3: Map of eastern United States depicting the mileages of streams located in the acid-sensitive areas of the homogenous regions. For these regions the stream mileages have been subdivided into sensitivity classes based on alkalinity distribution. Sensitivity classes are designated as follows: I-acidified; II-extreme; III-moderate; IV-low; V-not sensitive. (Circle size shows the magnitude of stream resource. Circles not subdivided indicate local akalinity measurements were not available for the regions to categorize the streams among the four sensitivity classes.)

value to estimate the proportion of the inventory that represents a modern "altered state" in surface water quality, as opposed to the proportion that has historically and naturally been in a low-alkalinity and often low-pH state. Failing the availability of precise alkalinity measurements on these lakes from a century ago, a putative "pre-alteration" inventory can be estimated by using data from regions of comparable soils and vegetation, but which are relatively free of modern atmospheric acid input.[4]

Using the results in Table 8 as representing the pre-alteration surface water alkalinity distribution (within probable error limits of ± 20%, equivalent to several percentage points), comparison with modern alkalinity distributions shows the proportion of the lakes and streams estimated to have undergone an alkalinity shift from one class to another. Lakes and streams that have experienced such a shift can be defined as "acid-altered surface water resources".

Tables 9 and 10 show the acid-altered resources in each of the 14 regions, obtained by subtracting the estimated pre-alteration resource for the

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[4] Such data are available for northwestern Ontario and northern Minnesota and were presented in Table 4. Less than one percent of the lakes there have alkalinities under 0, and less than 2 percent are below 40 ueq/l. These results are comparable to Adirondack lake alkalinities when one considers that observed negative alkalinities range down to -100 ueq/l (Pfeiffer and Festa 1980) and that declines in alkalinity there have been estimated at 100 ueq/l (Hendrey et al. 1980). Adding 100 ueq/l to Adirondack lake alkalinity distributions would result in alkalinity distribution between 0 and +100 ueq/l under pre-alteration times, similar to these of the modern Canadian Shield regions outside the heavily impacted areas. These results suggest the acceptability of northern Minnesota/N.W. Ontario lake data, and the northern Wisconsin stream data, as an estimate of the pre-alteration alkalinity distribution in acid bedrock, conifer/hardwood regions of the north-central, northeastern and Appalachian plateau regions of the eastern U.S. An independent estimate should be developed for the south central and coastal plain regions; but in view of the relatively small areas involved and the likelihood of a general similarity in historic values for these regions, the northern values are being used for the time being.

Table 8. Reconstructed historic or "baseline" number of lakes and miles of streams on a low-base igneous, metamorphic and sedimentary bedrock types, by sensitivity classes (see text).

	Sensitivity Class (alkalinity expressed in µeq/ℓ)				
Resource Type	Acidified ≦ 0	Extreme 0-40	Moderate 40-99	Low 200-499	Not ≥ 500
Lakes	1	2	36	26	35
Streams	0	2	11	33	54

Table 9. An estimation of the "acid-altered" lake resources of the eastern United States combined for alkalinity classes I and II, using 1980 calculated alkalinity depression and "historic" area alkalinity distributions as a control. (Estimated confidence, ± 20%.)

Region	Total No. Lakes	(1980) No. Lakes in Alk. Class I & II	Historic No. Lakes Alk. Class I & II	(1980) No. Lakes Altered Alk. Class I & II
1	1425	456	43	413
2	1543	494	46	448
3	1139	456	34	422
4	1320	423	40	383
5	376	150	11	[illegible]
6	13	----	----	----
7	126	----	----	----
8	241	----	----	----
9	170	----	----	----
10	9	----	----	----
11	583	134	17	117
12	5307	1220	159	1061
13	1637	49	49	0
14	3170	95	95	0
TOTAL	17059	3477	494	2983

Total number of lakes altered, Class I and II: 2,983

Percentage of lakes altered: 18%

Table 10. An estimation of the "acid-altered" stream resources of the eastern United States combined for alkalinity classes I and II and 38% of III[1] using both calculated alkalinity depression and alkalinity distribution from untreated areas as a baseline or control. (Estimated confidence, ± 20%.)

Region	Total Stream Miles	1980 Miles in Class I & II plus 38 % of III[1]	Historic Stream Miles in Class I, II + 38% of III	Stream Miles Altered, Class I, II + 38% of III
1	13,199	3,746	816	2,930
2	19,877	5,641	1,228	4,413
3	8,313	3,060	514	2,546
4	10,956	3,109	677	2,432
5	8,792	3,237	543	2,694
6	9,649	2,738	596	2,142
7	14,297	4,058	884	3,174
8	2,260	482	140	342
9	6,629	1,412	410	1,002
10	3,794	809	234	575
11	3,411	726	211	515
12	6,774	418	418	0
13	2,112	130	131	0
14	7,360	455	455	0
TOTAL	117,423	30,476	7,257	22,765

1. To correct for alkalinity distributions between streams and lakes, a defined portion of the streams in Class III (40 - 100 ueq/l) was included in the calculations.

2. Total stream miles altered, Class I, II and 38% of III: 22,765, representing 20% of the total first and second order stream resource.

two lowest categories from the modern inventory. After correcting for lakes and streams that were historically in a low-alkalinity state, about 18% of the total lakes and 20% of the total streams fall in the "acidified" (Class I) or "extreme" (Class II) sensitivity classes and are considered to be affected by acid deposition.

V. DESCRIBING AND MODELING SURFACE WATER pH AND ALKALINITY

The alteration of lake and stream water chemistry by wet and dry acid inputs appears to be controlled by two sets of interacting variables: geochemical and related physical/biological components of watersheds (which determine the neutralizing or buffering capacity of the watershed) and the mobility of acid anions (such as sulfate) in the deposition (Overrein *et al.* 1980, U.S./Canada 1981). For a given soil under natural conditions, the weathering of soil minerals appears to provide sufficient natural sources of neutralization capacity to maintain an approximate long-term equilibrium with natural sources of H^+. This equilibrium is normally acidic for northern conifer forests, but is nearly neutral for deciduous forests. Predicting precisely the consequences of moderate additions of acid for the equilibrium of these systems, or for the aquatic resources they affect, requires a model which accounts for both of the above sets of variables. Such a model requires detailed information on long-term trends, on basin geochemistry, and on soil biology that is only partially available at this time and is unlikely to be fully available soon. Until such data and models are available, however, an empirical model can be used to obtain first approximations of probable responses to increases or abatement of acidic inputs.

1. Time Trends of pH and Alkalinity in Lakes and Streams

Observed pH declines in lakes and streams in sensitive regions of Scandinavia and North America are summarized in Table 11. The average rate of decline (Δ pH/decade) is expressed here simply as the change in pH divided by the number of years over which it occurred, then multiplying by ten. However, such changes are not expected to have progressed at some constant rate. Change may have been very slow while the lake was reasonably buffered and rapid when the capability to neutralize acidity was depleted.

Hendrey *et al.* (1980) attempt to assess regional acidification in the Adirondacks, New Hampshire and North Carolina by analyzing observed changes in total alkalinity. In the Adirondacks, regional acidification of lakes, as measured by changes in alkalinity since the 1930's, is about 26 ueq/l per decade, Table 12. For North Carolina streams, the indicated decrease up to 1979 in relation to historical alkalinity data (1960-1963) is about 36 ueq/l (a decline of 9 ueq/l per decade). The results are summarized in Table 12 with reference to the regions defined in this study, and should be understood as applying only to the more sensitive streams or lakes, rather than to the entire population of aquatic resources. In general, the results are consistent with the magnitude of atmospheric SO_4^{2-} loadings to the regions shown.

Table 11. Observed changes in pH and the rate of pH declines for lakes and streams in sensitive regions of Scandinavia and North America.

Region	N	Historic pH	Recent pH	ΔpH/decade	Reference(s)
Lakes					
Central Norway	10	7.3(1941)	5.8(1975)	-0.44	Wright & Gjessing 1976
West Sweden	6	6.6(1933-35)	5.4(1971)	-0.32	"
	8	6.8(1942-49)	5.6(1971)	-0.48	"
West-central Sweden	5	6.3(1937-48)	4.7(1973)	-0.53	"
South-central Sweden	5	6.2(1933-48)	5.5(1973)	-0.21	"
Southern Sweden	51	6.8(1935)	6.2(1971)	-0.17	"
La Cloche Mtns., Ontario	7	6.3(1961)	4.9(1972-73)	-1.27	"
	8	5.0(1969)	4.8(1972-73)	-0.67	"
Adirondacks	8	6.5(1930-38)	4.8(1969-75)	-0.46	"
Maine	12	6.8(1937-47)	5.9(1974)	-0.30	Davis *et al.* 1978
Nova Scotia and New Brunswick	10	5.7(1950s)	5.3(1979)	-0.16	Wiltshire & Machell 1981
Streams					
New Jersey	2	4.6(1963)	4.1(1978)*	-0.30	Johnson 1979
North Carolina	38	6.8(1960-64)	6.5(1979)	-0.18	Hendrey *et al.* 1980
New Hampshire	68	6.7(1937-39)	6.1(1979)	-0.15	"
Nova Scotia	6	5.2(1954-55)	4.6(1973)	-0.32	Thompson *et al.* 1980

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* Average based on linear regression.

Table 12. Summary of regional rates of decline in lake and stream alkalinity in relation to long-term SO_4^{2-} loading (~ dose).

SO_4^{2-} Dose/Loading	Region	Field Estimate (μeq/ℓ/decade)
High		
(35+ kg/ha·yr)	Adirondack Lakes	26
	Hinckley Reservoir	39
Intermediate		
(20-35 kg/ha·yr)	New Hampshire	14*
	North Carolina	9
Moderate		
(10-20 kg/ha·yr)	Northern Wisconsin	No estimate
Low		
(10 kg/ha·yr)	Northwestern Ontario	No estimate

* Calculated using Henriksen model (Henriksen, 1980). There are no historical alkalinity data to check this estimate against.

2. Atmospheric SO_4^{2-} Input as an Important Factor

The time trends in pH, and the alkalinity decline described above, have occurred in generally sensitive areas, although some of the variability among the data may be due to local differences in soils and watershed properties affecting pH and alkalinity. Another causal agent for some of the observed difference in alkalinity, and its apparent rate of change, is the difference in cumulative acid dose, or its dominant anion surrogates, SO_4^{2-} (sulfate) and HCO_3^- (bicarbonate). Sulfate deposition data from the NADP and CANSAP networks have been presented by Glass and Brydges (1981), and are displayed as Figure 4.

The higher rates of pH and alkalinity changes in the Adirondacks and the La Cloche Mountains (noted above) are in regions of high SO_4^{2-} deposition, while the lower rates of change in North Carolina, New Hampshire, New Jersey and Nova Scotia (see Tables 11 and 12) are associated with intermediate rates of deposition.

Because most available nitrate is utilized by plants, at least during the growing season, nitrate deposition seems unlikely to have an appreciable effect on midsummer alkalinity data or trends in these data, although it may have an influence on spring pH depression.

The alkalinity distribution data already presented in Tables 3 and 11 appear to reflect an ongoing acidification process which has progressed farther in the eastern U.S. than in the upper Midwest. Considerable acidification has already occurred in Adirondack lakes. Indeed, the cumulative decrease in alkalinity is estimated by Hendrey et al. (1980) to

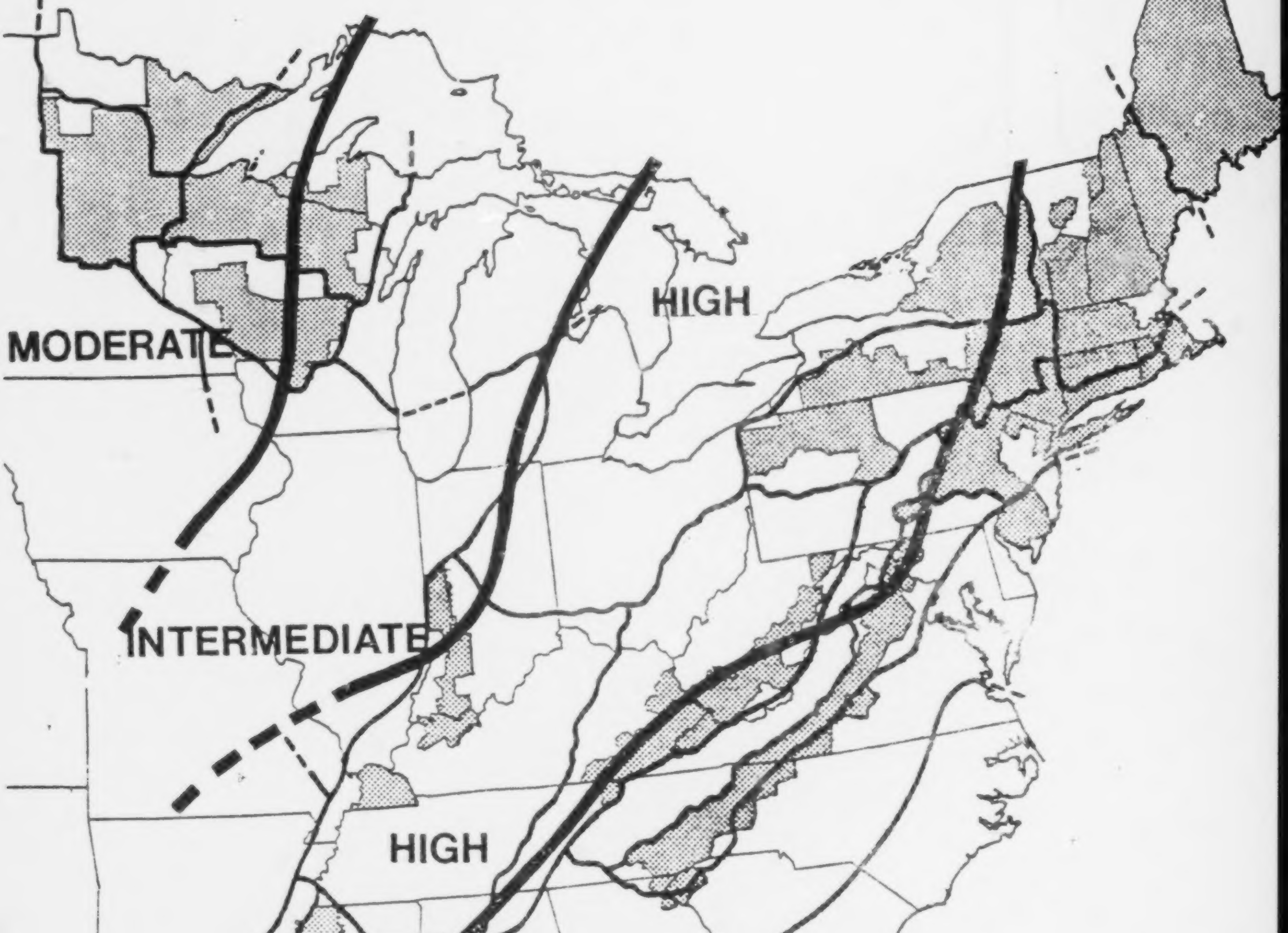
MODERATE
HIGH
INTERMEDIATE
HIGH
INTERMEDIATE
MOD.

Figure 4. Provisional distribution of SO_4^{2-} loadings in 1979-80 as an estimator of total cumulative dose: High (35+ kg/ha·yr), Int. (20-35), Mod. (10-20), and Low (<10).

total about 100 ueq/l. Thus, the 23% of lakes now reported to have acidified alkalinities from -100 to 0 ("acidified") would have had historical alkalinities in the positive range, but less than 100 ueq/l. The lakes now classified as "extremely sensitive" (Class II) were probably somewhat more alkaline.

Despite considerable research in Europe, Canada and the United States, no widely-accepted method is available for projecting future trends in lake acidification or recovery. Researchers appear to hold several viewpoints, of which two will be elaborated here. One states that the acidity and alkalinity of watersheds, streams and lakes normally comes to equilibrium with atmospheric acid inputs, and that either increases or decreases in atmospheric acid inputs would soon cause further changes in water quality (or biological effects). In this view, continuation of present emissions and regional weather and transport patterns would lead to no change in regional surface water acidification. Correspondingly, a reduction in emissions and deposition would lead to improved pH and alkalinity in some lake or stream systems, depending on local conditions.

A complementary view, emerging from recent interpretations of U.S., Canadian and Scandinavian data (Seip _et al_. 1979; Overrein _et al_. 1980; U.S.-Canada 1981, Johnson 1980, Morison 1981), holds that the transition from high-alkalinity (well-buffered) waters to acidified waters is based on cumulative processes in the watershed. In this view, acid-neutralizing elements in the soil and the soil's sulfur retention capacity are gradually depleted by long-term sulfate deposition. Once the saturation point is reached, sulfate anions must travel through the watershed to the water body, carrying hydrogen ions with them.

Although some scientists disagree, a combination of these two views is consistent with the evidence of continuing small changes in surface-water pH and alkalinity, despite little region-wide evidence of change in sulfate loadings during the past 10-12 years.

3. Fitting an Empirical Model

In creating a model to project changes in the status of aquatic resources, our objective is to relate changes in hydrogen ion concentrations (or alkalinity) in streams or lakes to changes in sulfate loadings to the watershed. One can approach the question empirically and try to define a parameter linking alkalinity changes in the water to sulfate and nitrate loadings on the land. An empirical "alkalinity impact parameter" (AIP) can be defined as the ratio of observed changes in lake and stream alkalinity levels to changes in sulfate loadings to the watershed.

When: 1. Sulfate retention by the watershed approaches zero, (2) nitrate uptake by plants is high, and (3) further losses of nutrient cations -- for example, calcium and magnesium -- are low or negligible, H_2SO_4 acidity directly affects alkalinity (HCO_3^-), and the expected AIP is 2.[5] For the case where some buffering of acid input occurs in the watershed, the AIP would be less than 2; during the very early stages of an acidic deposition it should approach zero. For the case where some SO_4^{2-} retention occurs, the AIP should approach 0 during "initial acidification phase" (adsorption) process, but could result in values greater than 2 during the "recovery phase" as some of the SO_4 adsorbed leaves the watershed and passes into surface waters.[6]

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[5] In this case, a direct response, HCO_3^- (in ueq/l yr) per unit SO_4^{2-} (in kg/ha yr) yields an expected AIP of 2 (ueq HCO_3^-/kgSO_4^{2-} ha yr), assuming 100 cm of rainfall with the acidic deposition.

[6] Limited data from three sites (Hubbard Brook, Hinckley Reservoir, and Nova Scotia Rivers) indicate AIP values could exceed 2 in these areas.

A computational procedure for calculating the change in the alkalinity of surface waters associated with a change in SO_4^{2-} loadings has been devised and is described in the full report. The critical alkalinity impact parameter has been set at two different values in the computations (AIP of 1 and 2)[7], reflecting a moderate range of cation exchange status, and forest soil CO_2 concentrations. A value of 0 implies no change.

Since an empirically-estimated property such as the AIP is expected to vary spatially from one watershed to another, and over time as soil pH (if altered) responds and affects the HCO_3^- equilibrium, simple integer values over the range of 0 to 2 are all that should be used at this time. The continuous spatial and temporal variability of this parameter should be investigated as an expansion of research on the three dominant acid-mediation processes in watersheds.

VI. FUTURE TRENDS IN LAKE AND STREAM ACIDIFICATION

1. Scenarios of Future Sulfate Deposition

Three scenarios of sulfate deposition from precipitation through the year 2000 were considered in this report:

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[7] The alkalinity impact parameter is expressed in units of: (ueq/ℓ HCO_3^-)/(kg/ha.yr SO_4^{2-})

1. A 10% increase in sulfate deposition by 2000.
2. A 20% decrease in sulfate deposition by 2000.
3. A 35% decrease in sulfate deposition by 2000.

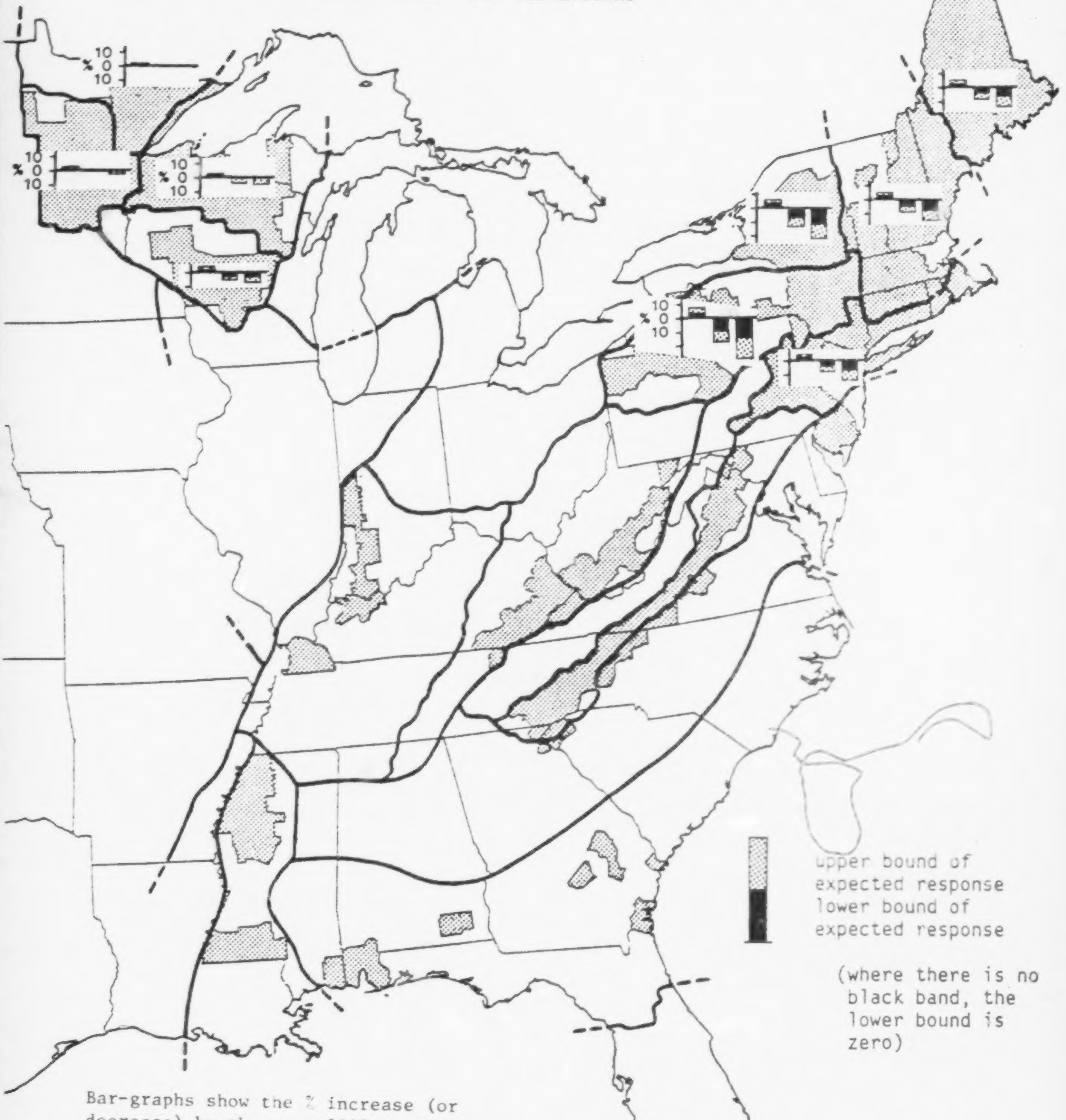
For simplicity, the changes in deposition were applied as a uniform proportionality to each of the 14 regions. 1980 baseline deposition values were obtained from the NADP and CANSAP networks (Glass and Brydges, 1981).

2. Results of Computed Change in Acidification Status by 2000

Using the empirical alkalinity response model described in the previous section, and the estimates of SO_4^{2-} loadings for three scenarios for the year 2000, projected changes in lake and stream alkalinity distributions can be computed. The computation procedure requires an assumption that the lake and stream alkalinity data can be represented as two populations evenly distributed over the ranges shown for Class I (to -40) to Class V, and that a projected increase or decrease in the alkalinity of these resources can be represented by shifts to adjacent classes.

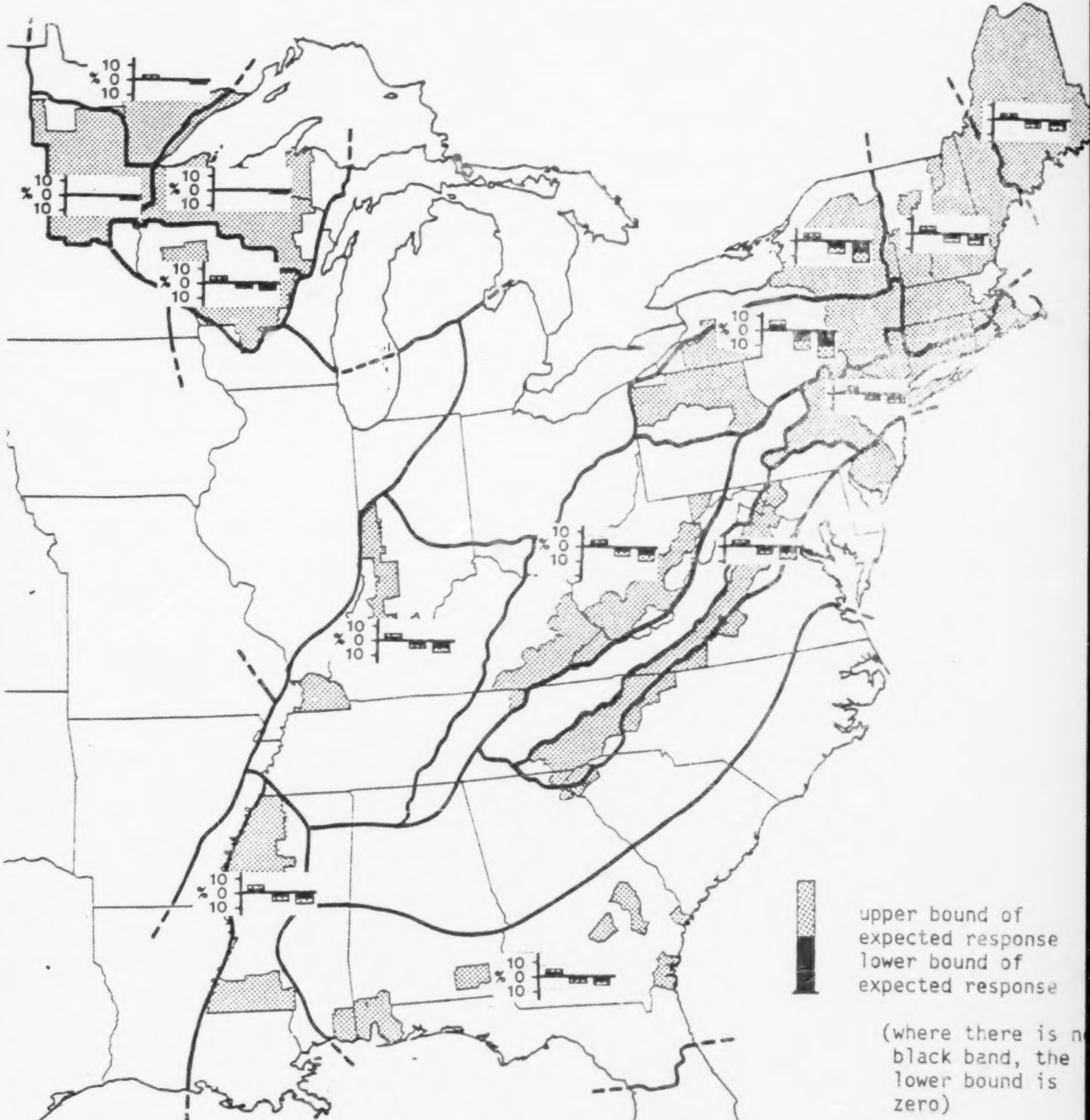
Many approaches to summarizing the model results are possible. One view of the general effect of the SO_4^{2-} deposition trends postulated in the three scenarios is summarized in Figures 5 and 6. Scenarios 1, 2, and 3 are shown in that order in each bar-graph. In these two Figures the change being shown is the sum of the calculated shifts to or from either of Classes I and II, expressed as a percent of the total lake or stream resource in each area. Because of the preliminary status of the model and its limited validation to date, a broad range of results must be presented and interpreted. The short, solid portion of each bar represents a highly probable response (with the AIP at a value of 1); the shaded portion of each bar represents the response expected with an AIP of up to 2.

FIGURE 5. LAKE MODEL PROJECTIONS


% 10 0 10
% 10 0 10
% 10 0 10
% 10 0 10
upper bound of expected response
lower bound of expected response
(where there is no black band, the lower bound is zero)

Bar-graphs show the % increase (or decrease) by the year 2000 in lakes classed as "acidified" or "extreme" for the three deposition scenarios I (a 10% increase in deposition), II (a 20% decrease), and III (a 35% decrease), represented by the bars, in that order from left to right.

FIGURE 6. STREAM MODEL PROJECTIONS


10
%
0
10
upper bound of expected response
lower bound of expected response
(where there is n
black band, the
lower bound is
zero)

Bar-graphs show the % increase (or decrease) by the year 2000 in streams classed as "acidified" or "extreme" for the three deposition scenarios I (a 10% increase in deposition), II (a 20% decrease), and III (a 35% decrease), represented by the bars, in that order from left to right.

For both lakes and streams, a 0-7% increase (depending on the region) above the present number of Class I and II lakes and Class I, II, and 38% of III streams ("acidified" and "extreme sensitivity") is projected for the 10% increase in sulfate deposition under Scenario 1. For Scenario 2 (20% reduction in SO_4^{2-} loadings), recovery appears likely to affect 0 to 18% of the Class I and II lake resources (including the shaded bars) and from no change to over 10% of the stream resources.

The responses under Scenario 3 by the year 2000 (a 35% reduction in SO_4^{2-} loadings) appear likely to be relatively significant. Recovery is projected to range from a few to about 30% of the lakes in each area and up to 20% of the streams, with the greatest recovery in areas of highest current deposition.

It should be noted that a transfer from Class I to Class II does not represent complete recovery of the resource, and, since the percentages shown here represent resources shifting between classes, one can say only that the apparent effects represent a substantial recovery of alkalinity distributions. In the western and southern regions, the model assumptions of the status of the soil do not hold well, and therefore the Alkalinity Impact Parameter recovery estimates shown here may be high, even at an AIP of 1.

A somewhat different approach to summarizing the projected effects of Scenarios 1, 2, and 3 on surface water resources is presented in Tables 13 and 14. Here, a numerical range for the number of lakes and miles of stream expected to undergo a substantial change by the year 2000 is presented for each region and each scenario. A "substantial change" is defined again as a shift from one alkalinity class to another, with each of the relevant classes (or parts thereof) having a range of 40 ueq/l. A parameter value of 1.0 is used as a lower bound, and a value of 2.0 as a maximum for regions 1 to 10;

Table 13. Estimated total changes in the status of lakes in alkalinity Classes I and II by the year 2000 in relation to total acid-altered lake resources in 1980 for Scenarios 1, 2 and 3.

Region		No. Acid Altered Lakes, 1980	Change in no. altered lakes, 2000		
			Scenario 1 (Increase)	Scenario 2 (Decrease)	Scenario 3 (Decrease)
1	Eastern Maine	413	19 - 38	57 - 114	91 - 182
2	Western New England	448	31 - 61	62 - 123	111 - 222
3	Adirondacks	422	31 - 62	80 - 159	137 - 273
4	East Pennsylvania S. New England	383	26 - 52	53 - 106	95 - 190
5	W. New York/Pennsylvania	139	13 - 25	34 - 68	56 - 113
11	Central Wisconsin	117	0 - 7	0 - 13	0 - 20
12	Wisconsin/Michigan Highlands	1,061	0 - 66	0 - 153	0 - 214
13	Northeast Minnesota	0	0 - 9	0 - 4	0 - 6
14	Central Minnesota	0	0 - 9	0 - 5	0 - 10
		2,983	120 -329	287 - 745	490 -1230
Percent of Acid-altered Lakes:			4 - 11	10 - 25	16 - 41

Table 14. Estimated total change in the status of 1st and 2nd order stream miles in alkalinity Classes I, II and 38% of III, by the year 2000 in relation to total acid-altered stream resources in 1980 for Scenarios 1, 2 and 3.

Region	1980 miles of altered streams Class I, II and 38% of III	Change in miles of altered streams, 2,000		
		Scenario 1 Increase	Scenario 2 Decrease	Scenario 3 Decrease
1	2,930	147 - 295	228 - 457	365 - 731
2	4,413	333 - 666	344 - 688	619 - 1238
3	2,547	184 - 369	374 - 748	641 - 1282
4	2,432	183 - 367	190 - 379	341 - 683
5	2,693	244 - 487	508 - 1017	847 - 1695
6	2,142	215 - 431	234 - 468	401 - 802
7	3,174	239 - 479	297 - 594	495 - 990
8	342	20 - 40	32 - 64	51 - 102
9	1,002	117 - 234	150 - 300	263 - 526
10	575	67 - 134	75 - 150	140 - 279
11	515	0 - 30	0 - 39	0 - 58
12	0	0 - 22	0 - 26	0 - 36
13	0	0 - 7	0 - 5	0 - 8
14	0	0 - 12	0 - 11	0 - 22
	22,765	1749 - 3573	2432 - 4946	4163 - 8452
Percent of Acid-altered Streams:		8 - 16	11 - 22	18 - 37

for regions 11 to 14, 0 is taken as a minimum and 1.0 as a maximum.

The resource responses (positive or negative change) are considered here in light of the historic or baseline alkalinity distributions shown in Tables 3 and 4, and the "acid-altered" resources are defined by the difference between historic and modern observations. As indicated previously, these acid-altered resources are roughly equivalent to the "acidified" and "extreme" alkalinity/sensitivity classes, differing only in subtraction of an historic baseline percentage of very low-alkalinity lakes and streams. Because of the generally higher alkalinities of stream resources (approximately 60 ueq/l, see discussion of Tables 3 and 4), a portion of the transfer into and out of alkalinity Class III (with a 160 ueq/l range) has been included for streams (i.e., 60/160, or 38%) to maintain a parallel basis for comparisons between lakes and streams.

The results in Tables 13 and 14 show that in the most heavily impacted areas of the northeast, several hundred lakes and several thousand miles of stream appear likely to recover substantially by the year 2000 under both of the major abatement scenarios. New negative impacts on surface water resources would be moderate under the scenario reflecting a 10% increase in deposition. Considering the total change in all 14 regions, a minimum loss of 120 additional lakes (representing a 4% increase in the acid-altered resource) and a more probable loss of 330 lakes (representing an 11% increase in the acid-altered resource) can be expected with the increased SO_4^{2-} deposition under Scenario 1. For streams, the likely minimum and probable losses are 1750 and 3570 miles, respectively, representing 8 and 16% of the resource respectively.

A moderate degree of recovery is projected under Scenario 2 results. The total change (recovery) projected for lakes here is 290 to 750 lakes, representing 10 to 25% of the total lake resource. For streams, the results are 2430 to 4950, or 11 to 22% of the resource. With Scenario 3, the levels of recovery projected are almost double those of Scenario 2, across the board.

3. Discussion

The projected recovery of surface water resources shown in Tables 13 and 14, although presented as a broad range, must be viewed as an estimate within a background of more uncertainty than is usually tolerable. These results ignore the recovery of surface waters (following decreased sulfate deposition) from flushing of accumulated sulfate and from decreases in cation removal rates, both of which are implicit in Alkalinity Impact Parameter values above 2. However, only a few regions can be considered to exceed an AIP of 2 (certainly not those of the upper Great Lake States), and the simplest approach has been to present only the more conservative summary.

Despite the indications here of some important prospects for recovery, the model cannot address some relatively irreversible components of acidification. If soil nutrients have been depleted over a long-term, nutrient levels will not recover as much as the other results suggest. Thus, incomplete buffering of watersheds (and toxic shock episodes) may continue in the systems shown here as "partially recovered", with consequences of a magnitude that are unpredictable at this time. Flushing of the SO_4^{2-} is also likely to continue over an extended period; some recovery of percent base saturation in soils is possible over a few decades, but no predictions about these processes can be made here.

Several other important limitations must be attached to these results. Validation studies regarding the basic relationships between available alkalinity input data and the AIP values are incomplete at this time.

Although more information is becoming available to evaluate the model in conjunction with other watershed and lake processes (such as sulfate retention and cation stripping), additional field data are needed. Because of the uncertainty in estimating the AIP, the reliability of the present results may be only within ± 50%, either in the heavily-acidified north-central and northeastern regions, or in the Midwest and South.

Finally, a comment is needed on the implications of these results for biological responses in aquatic systems. Scientists are rapidly developing an understanding of the relationship of fish and other aquatic life to the chemistry of their surrounding environment. (See Overrein 1980, Harvey _et al_. 1981 and others). Thus, predictive statements about changes in water quality measures over some period of abatement represent the principal means of making predictive statements about the potential recovery of the biota. Other issues are also important, however, and another important question is whether biological productivity in recently-acidified systems can recover over a period comparable to the recovery of water quality.

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Appendix JJ

SECTION I HEALTH EFFECTS OF SO_x

1. INTRODUCTION

The complex mix of air pollutants from fuel combustion poses a variety of threats to man and his environment. Many pollutants present in combustion emissions are known to produce acute health effects at high concentrations. Chronic health effects from the low concentrations typical of normal exposures are more difficult to demonstrate.

Although an association of air pollution with health damage has been demonstrated, there are unresolved questions about which particular pollutants cause the damage and how they operate. Sulfate concentrations are correlated with mortality, but whether they are a causative agent or an indicator of other processes is a controversial question. Partitioning of sulfur emissions into short-range and long-range components, and chemical transformation from SO_2 to SO_4, require further definition. Accordingly, this analysis first examines potential health effects of local air pollutants, for which exposures can be high and responses are easily demonstrated. It then discusses potential chronic health effects from low-level exposures to pollutants transported over long distances.

2. DEGREE OF KNOWLEDGE

Despite many years of multidisciplinary study, specific health effects of air pollutants are not known with a high degree of certainty. Problems in measuring exposures, plus difficulties in investigating human effects at low exposures or in relating animal data to humans, combine to yield large uncertainties in dose-response relationships and in mechanisms of damage. In addition, predictive estimates suffer from uncertainties about locations of pollution sources; transport, transformation, and deposition of pollutants; and demographic variables affecting total exposure.

3. LOCAL AIR POLLUTANTS

Concentrations of air pollutants can be high around local emission sources. The best-known local air pollutants are SO_2 and CO, which generally disperse to concentrations that are not of concern beyond the immediate vicinity of the source, or undergo chemical reactions to form other compounds. Large (>10um) particles have traditionally been a local air pollution problem. Fine-particle emissions of organics, directly-emitted sulfate particles, etc., can be important locally as well as over long distances.

The distinction between local and long-range pollution involves two elements: 1) the location of high-concentration areas relative to sources; 2) chemical characteristics of the pollutants and their dose-response relationships. Pollutants that degrade quickly or change form, like SO_2, and those with high health effects thresholds, are more likely to be of significance locally where concentrations are higher. Secondary products of chemical reactions in the atmosphere, e.g., ozone and sulfates, are more likely to be found far from emission sources.

3.1 Sulfur Dioxide

Inhaled SO_2 does not penetrate deeply into the lung before being absorbed and incorporated into the secretions lining the respiratory passages. Animal studies of SO_2 exposure show an immediate but transient broncho-constrictive effect from single high doses (Amdur, et al., 1975). SO_2 is potentially mutagenic in microbial test systems (Fishbein 1976) and limited animal studies have suggested that SO_2 may be a co-carcinogen in rats, enhancing the carcinogenicity of benzo(a)pyrene (Laskin, et al., 1969).

SO_2 is generally not thought to be a significant health risk to normal persons at normal environmental concentrations. Recent clinical evidence has shown, however, that low-level SO_2 exposure can cause significant changes in lung function (bronchoconstriction) in asthmatics (Sheppard, et al., 1981). The effect was seen during mild or moderate exercise while exposed to 0.5, 0.25, and, in two sensitive asthmatics, 0.1 ppm. of SO_2. (The 24-hr. federal standard is 0.14 ppm.) It can be inferred that transient peak concentrations of SO_2 in ambient air at or above the present 24-hr. standard are capable of inducing similar functional changes in asthmatics as well as in others whose airways may be in an irritable state. These acute functional changes are completely reversible. Chronic exposures have been shown to decrease particle clearance rate and to increase susceptibility to infection (Hirsch, et al., 1975).

4. LONG-RANGE AIR POLLUTANTS

4.1 Sulfates and Other Aerosols

The primary problem with sulfur pollution is not thought to be SO_2 itself, but the sulfate products that form during chemical reactions with other substances in the atmosphere, and the interactions between SO_2 and other harmful particles. SO_2 is oxidized in ambient air to sulfuric acid (H_2SO_4), a strong acid sulfate (pH $\leq$ 1). H_2SO_4 can then be neutralized partially to ammonium bisulfate (NH_4HSO_4, pH 1-2), or completely to ammonium sulfate [$(NH_4)_2SO_4$, pH 5-6] by trace amounts of gaseous ammonia (NH_3). All three molecular forms of sulfate are present as fine particles, i.e., largely under 1.0 um diameter at ordinary relative humidity. These particles reach deeper into the lung than SO_2 gas and are retained longer before they are cleared and swallowed. Retention periods can be several hundreds of days.

Sulfuric acid (H_2SO_4) is the strongest irritant sulfate (Table 1). When inhaled, as demonstrated in both clinical and animal experiments, H_2SO_4 has two important effects on the lung: (1) bronchoconstriction, which increases the work of breathing (Amdur, et al., 1952), and (2) a change in the rate at which the lung clears itself of inhaled particles, including dusts and microbes (Bushtueva, 1957; Schlesinger, et al., 1979; Leikauf, et al., 1981). If lung clearance is impaired, susceptibility to local infection, chronic disease, and cancer is likely to increase.

Fine particles can also adsorb polycyclic organic matter, some of which (benzo(a)pyrene, for example) is known to be carcinogenic. Again because of their small size, the carcinogen-carrying particles penetrate deep into the lung where they can remain for many weeks. Not only are both carcinogens and sulfates potentially harmful in their own right, but there is also evidence that the tissue irritation produced by acid sulfates (and SO_2) can make lung tissues more susceptible to the carcinogenic effects of the polycyclic organic matter (Peacock and Spence, 1976).

Populations at special risk of health damage from particulate sulfates include the elderly and adults with pre-existing chronic heart or lung disease. Children also appear to be susceptible to increased lower respiratory tract illness and more transient effects such as decreased lung function. Asthmatics of all ages are especially sensitive to lung irritation from all sources.

In spite of the potential for sulfate particles to cause damage, the direct evidence available indicates that they have no detectable effects at likely environmental concentrations. [1] Nevertheless, it is clear that something in the sulfate-particle mix is harmful to human health. Substantial evidence has been gathered over more than 30 years indicating adverse health effects of sulfur-particulate air pollution. This includes evidence of increased mortality during short-term episodes [e.g., London 1952 and 1962 (Ministry of Health 1954) and Donora, Pennsylvania (Schrenk 1949)]; time series analysis over longer periods [Oslo, Dublin (Lindberg 1968)]; cross-sectional analysis among cities of different air pollution levels [e.g., Japan (Nishikawa, et al., 1979)]; and cohort epidemiology [Berlin, NH (Ferris, et al., 1979)]. Pollution was measured in terms of a specific index (SO_2, TSP, sulfation rate, etc.), but it has generally been impossible to attribute effects to any single part of the entire pollutant mix. Nevertheless, evidence was felt to be strong enough, coupled with other reasons (e.g., aesthetics), to justify undertaking massive air pollution control programs. In London, emphasis was placed on reduction of particle levels -- with an apparent effect on health. In New York City, emphasis was on SO_2 (although particles were also reduced somewhat), without any evident effect on health (Schimmel and Murawski, 1975).

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[1] Avol, et al., 1979; Schlesinger, et al., 1978; Wolff, et al., 1979.

In the early 1970's, a combination of evidence seemed to point to sulfate particles, particularly acid forms, as the health-damaging agent in the sulfur-particulate mix. Toxicological studies of animals (Amdur, 1977), epidemiological studies [2] (EPA 1974), and correlational studies of general health statistics (Lave and Seskin 1977) all pointed in this direction. Even at that time, however, the evidence was not judged sufficient for the usual criteria of scientific "proof" (Morgan, et al., 1978), and some investigators were unwilling to accept the concept at all (Morris and Morgan 1974).

In addition, some of the epidemiology linking health damage with sulfate has come into question (Committee on Sci. and Technol. 1976). Recent work in air chemistry has shown that, while some free sulfuric acid aerosol exists in the atmosphere, the average acidity of the particles is close to that of ammonium bisulfate (Stevens, et al., 1980). The latter has been shown in animal studies to be of lower potency than sulfuric acid and some metallic sulfates, although still higher than SO_2 gas (Amdur, 1971).

Therefore, no one pollutant can be specifically identified as the primary agent. Sulfate remain a candidate, however, and still may be the best overall index of health impact. Most probably it is the combination of sulfates and other irritants acting synergistically with other particulate pollutants that causes the observed effects. Leikauf and Lipmann (1980) conclude that at ambient air levels ". . .there is no clear evidence that the United States population is currently suffering any direct health effects from the inhalation of various sulfur oxide compounds [but] our current knowledge of mechanisms of action, dose-response, and possible interactions with other pollutants is so poor that we clearly have no basis for complacency either." Much current research identifies submicrometer sulfate aerosols, particularly acidic species, to be of special concern.

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[2] Epidemiological studies involve statistical analyses of relationships between incidence of disease and prevalence of possible causes.

Despite lack of firm evidence, policy studies require some attempt to estimate health effects of air pollution. Clearly there is no justification for assumptions that air pollution has no health impacts. Therefore a health damage function based on sulfates has been developed which explicitly incorporates the uncertainty in knowledge (Morgan, et al., 1978). While controversial, the approach has been widely used in both the U.S. (Morris, et al., 1979) and Europe (Hamilton, 1980). This damage function is used below to estimate health effects of pollution emissions. Sulfate is taken as an index of the sulfur-particulate air pollution mix and of the health effects of long-range pollution transport. Its use in this manner remains controversial, but it is the best we can do with the current state of knowledge.

4.2 Level of Agreement Among Researchers

There is fairly general agreement among researchers that levels of exposure exceeding air quality standards [3/] -- above a combined annual average of about 100 ug/m^3 TSP and about 100 ug/m^3 SO_2 -- or extreme episodic conditions, cause adverse health effects. Disagreement about the effects of low levels of exposure falls into two classes:

[3/] 75 ug/m^3 TSP, 80 ug/m^3 SO_2 annual average.

(1) At low levels of exposure -- near and below current TSP and SO_2 air quality standards -- there is no evidence of health effects from epidemiological studies of individuals, from clinical studies, nor from animal studies. Effects must be estimated from correlational studies of general health statistics which are subject to considerable error from misspecification and multicausality, and from extrapolation from higher doses. Some believe there are threshold levels which preclude extrapolation from higher doses, and that there are no effects below such levels; these thresholds may be related to neutralization of acid sulfates by ammonia in breath, (Larson, et al., 1977), or to other natural defense or repair mechanisms. Others allow for the possibility of low-level effects.

(2) Some researchers do not consider existing information sufficient to say anything about which agent(s) within the sulfur-particulate mix may be hazardous. Others are willing to accept indices such as sulfates as a means to estimate health impacts, taking into account the considerable uncertainty in such use and the difficulty in using these indices to evaluate energy policies or pollution-control strategies.

4.3 Acute vs. Chronic

At high levels of exposure (in excess of 200 ug/m^3), sulfur-particulate air pollution can aggravate existing conditions such as asthma, chronic bronchitis, and heart disease. It is fairly well accepted that those who died in smog episodes were people with pre-existing cardio-respiratory disease. (Note that such people were not necessarily about to die anyway: a severe bronchitic might live a restricted but long life in the absence of an air pollution episode.) In less extreme episodes, many of these effects can be transient; e.g., temporary increased discomfort. Transient effects can be quite serious, however, causing increased emergency room visits, for example.

The well-known severe air pollution incidents provide considerable evidence of acute episodic effects. Careful statistical studies of daily mortality in New York City have led to better insights into the effect of weather on such episodes (Schimmel and Murawski, 1975). More importantly, these studies demonstrate a continuum: small increases in daily air pollution are associated with small increases in daily mortality and large increases with higher mortality.

There is also considerable evidence that sulfur-particulate air pollution causes increased acute respiratory disease in children (Douglas and Waller, 1966), although some of the evidence is conflicting (Shy, et al., 1973).

Evidence for chronic effects is not as strong. Nevertheless, mortality studies are consistent with a hypothesis that air pollution causes increased chronic respiratory disease and that there is a cardio-respiratory relationship leading to heart disease and early death. A large number of studies -- most from the early literature -- show increasing prevalence of chronic respiratory disease with increasing sulfur-particulate pollution. Some of these control for smoking, socio-economic effects, etc., but most do not. Many similar studies show that increased sulfur-particulate air pollution is linked with decreased pulmonary function, as measured by several standard tests.

It has been hypothesized that increased acute respiratory disease in children leads, with continuing air pollution exposure, to increased chronic respiratory disease in adulthood (Holland, 1972). The link between childhood disease and later chronic bronchitis is weak. Follow-ups on two British studies of children have been unproductive (Lunn, et al., 1970; Colley, et al., 1973).

Current concern is with effects of long-term exposure to comparatively low levels of pollutants, some of which can be significantly affected by long-range transport. The evidence of effects at such exposures is controversial and depends strongly on extrapolation from studies at higher levels or on studies which have great potential for error.

Because particulate sulfates are irritants, their primary health effects tend to be mostly transitory and of light severity. Chronic irritation, however, can lead to permanently reduced lung function, lung disease, and heart disease from reduced lung function. The more severe effects presumably lead to premature death. The numbers of persons experiencing each level of health effect decreases with increasing severity. A large portion of the total health impact of particulate sulfates is morbidity -- so low in severity that many persons may not even be aware of them. Because most of the cases are never reported, the morbidity statistics available for epidemiological analysis are of exceedingly poor quality, where they can be found at all. For this reason, epidemiologists have relied on mortality statistics, the quality of which is far superior. A dose-response function based on mortality effects presumes that for every person dying prematurely, a large number of persons are sick, but either do not die, or die of causes unrelated to pollution exposure and therefore with no obvious relationship to the exposure data. For sulfates, it has been estimated that about 5 persons are nonfatally ill for every person who dies prematurely (Hamilton, 1980).

Lave and his co-workers Seskin (1970, 1973, 1977, 1978) and Chappie (1980) analyzed the relationships among urban mortality rates, air pollution concentrations, and socio-economic variables in 1959-61, 1969-70 and 1974. Sulfate particles and total suspended particles (TSP) were found to be strongly associated with increased mortality. These analyses have been examined in detail over the course of the past decade, and others have carried out studies using similar data with varying results (Lipfert, 1978; Hickey, et al., 1970; Schwing and McDonald, 1976; Crocker, et al., 1979). Studies by Bozzo have treated sections of individual cities (Gregor, 1977).

These studies are observational rather than experimental; they examine populations going about their usual business. As such, they suffer from a number of inherent inadequacies: they cannot distinguish between true causative factors and other associated variables (multicollinearity); nor can they determine whether important variables have been omitted. There is neither randomization nor other means of control, and aggregate statistics are used rather than data on individuals. Their cross-sectional design is appropriate for study of chronic -- not acute -- disease effects.

Less fundamental but equally important shortcomings also exist. One is the difficulty of accounting for the cumulative exposure of populations subjected to a history of varying pollution levels, some of whose members have migrated from locations with different pollution histories. Another, common to most air pollution epidemiological studies, is that measured outdoor pollutant levels can be quite different from actual exposures for populations that spend most of their time indoors. Further, these studies must rely on indices of a total pollutant mix; yet the relationship of an index pollutant to the total mix varies from place to place.

Finally, since the studies deal with aggregate population statistics rather than data on individuals, the distribution of air pollution exposure and the detailed effects of other variables at the individual level cannot be known -- although studies on sub-areas of a single city narrow the aggregation problem. The basic design of the studies is thus not useful for discovering mechanisms of effects, and may in fact mask them.

The virtue of such studies is that they make direct statements about human health: if a valid relation is found, then ipso facto human health is affected. One need not infer hypothetical health effects in man by analogy with damage observed in another species, nor extrapolate from small selected samples of human populations (often restricted to specific classes of individuals such as young, healthy workers) to the general population. Detailed epidemiological studies of randomly-selected individuals are much preferable, but -- because of cost and difficulty -- have been limited to small sample sizes.

Observational studies in general are unable to detect small health damage increments over pre-existing and indistinguishable diseases, but they are extremely useful for detecting larger effects, for setting upper boundaries on the effect of low-level exposures, and as a means of gaining further understanding of exposure patterns among populations and mechanisms of air pollution damage. For a more detailed discussion of the Lave-Seskin methodology results, see Appendix JJ-A.

4.4 Dose-Response Function

Some researchers maintain that because of the above-mentioned uncertainty, these studies cannot provide reliable evidence for estimating health damage from air pollution. Classic statistical theory does not provide a means of estimating uncertainty under these circumstances. To surmount this problem, Morgan et al. describe the uncertainty of the sulfate health damage coefficient in terms of a subjective probability distribution (see Figure 1) based principally on the range of results from alternative formulations, including those of the Lave-Seskin critics who find no effects.

We use sulfate exposure and the Morgan et al. sulfate dose-response function as a surrogate for the health effects of the sulfur-particulate pollution mix. The results center on the range of the Lave-Seskin average sulfate coefficients. There is a considerable probability of a lower effect, however, and a 14% chance of zero effect. The lower effect levels include the possibility that sulfates not only are not harmful to health, but also are not correlated with other pollutants which are harmful. This would not indicate that the sulfur-particulate pollution mix is not harmful, but rather that sulfates are not a good surrogate -- so that the distribution of pollution health effects cannot be predicted from sulfate concentrations.

Use of linear, non-threshold dose-response functions for pollutants other than carcinogens can be criticized as unrealistic. Many pollutants, particularly irritants, are likely to have physiological thresholds -- exposure levels below which the body's natural defense mechanisms are able to counteract their harmful effects. While this may be true for most pollutants

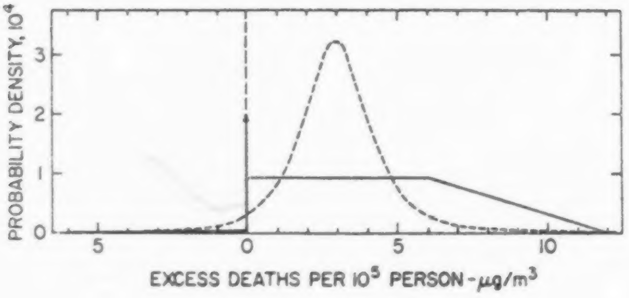

Figure 1. Subjective probability function (solid line) representing a composite of estimates for sulfate health damage. A delta-function at the origin containing 14% of the total probability represents the possibility that sulfate damage is zero, while the small negative tail (1% of the total probability) represents the inability of regression results to exclude the possibility of beneficial effects. The dashed curve is the classical estimate (Student's t distribution with three degrees of freedom) obtained by treating the results of four other regression studies as independent observations of the same rate. Source: Morgan et al., 1978:300.

and most individuals, there is always a range of responses to any potentially harmful exposure. A population-average dose-response function must, therefore, extend to zero to include those few especially sensitive individuals (Carnow, 1974). It can similarly be argued that linear functions, with or without thresholds, are unrealistic, because the variability of responses insures that only a few individuals will be sensitive at low exposures, but many will be sensitive at higher exposures, producing a non-linear function which rises relatively rapidly at exposures near the "average threshold" and reaches a maximum at exposures which affect virtually everyone. If this is true for the sulfate-particulate mix, the linear dose-response used in this report would underestimate mortality effects.

Use of sulfates as a surrogate or indicator of effects of a mixture of pollutants further removes the dose-response function from toxicological responses of specific individuals to specific pollutants. Each component of the pollution mix probably has a different threshold, if any, and a different effect, the combination of which is represented by sulfate exposure alone. In addition, threshold responses are likely to be related to short-term exposures to relatively high concentrations. Such concentrations generally occur in small areas and are not well quantified by large-scale (i.e., state-level) annual average concentrations. Given these confounding factors, both the nature and the scale of the surrogate dose-response function clearly preclude establishing all but the simplest, linear non-threshold relationship.

Readers should keep in mind that use of mortality attributable to pollution exposure does not necessarily imply that the mortality impacts are the most significant. As pointed out above, the mortality dose-response function is used because the available data on morbidity are inadequate for detailed epidemiological analyses. It has been estimated that, for every person dying from sulfate exposure, five persons are ill with lung disease, heart diseases aggravated by lung dysfunctions, and respiratory infections aggravated by lung irritations (Hamilton, 1980).

5. HEALTH IMPACTS OF LONG-RANGE TRANSPORTED POLLUTANTS IN THE UNITED STATES AND CANADA

Because long-range transported pollutants are distributed over large areas about emission sources, and because population density is not uniform throughout these areas, the distribution of human exposures and resulting health impacts can differ significantly from the distribution of emissions. As a result, there can be inequities in which persons bearing the health burden produced by the emissions do not also accrue the benefits associated with production of those emissions. Such inequities can be interstate and international.

5.1 Scenario Analysis

Three SO_2 emission scenarios were analyzed: (1) a 1978 base case for total SO_2 emissions from the United States and Canada; (2) a 2000 projection in which total SO_2 emissions are held constant at 1978 levels; and (3) 2000 rollback projection in which total SO_2 emissions from all states and provinces are linearly reduced by 30% from 1978 levels. The 2000 projection using 1978 emission levels represents a modest increase in projected emissions, while the 30% rollback from 1978 emission levels represents a modest decrease in projected emissions from the "business as usual" projection in the 1978-2000 interval.

5.2 Methods

5.2.1 Sulfur Dioxide Emissions

All SO_2 emissions were obtained from E. H. Pechan and Associates, Inc. Base case emissions for total SO_2 from all sources were assumed to be those from 1978 as catalogued by the U.S./Canadian Work Group on Transboundary Pollution (1981), supplemented by data from the U.S. Environmental Protection Agency's National Emissions Data System (NEDS) for 1978 (Pechan, 1981).

SO_2 emissions projections for 2000 included a "high" emissions scenario using 1978 emissions levels unchanged, and a "low" emissions scenario in which 1978 emissions levels were uniformly rolled back 30% from 1978 levels. Figures 2 through 9 show distributions of emissions and emissions densities. The emission pattern for the 30% rollback scenario is identical to the baseline, but with a 30% reduced magnitude.

5.2.2 Long-Range Transport and Transformation of Sulfur Dioxide to Sulfates

All SO_2 transport and transformation data were obtained from E. H. Pechan and Associates.

State- and province-level sulfate concentrations were estimated using an SO_2-SO_4 transfer matrix generated by the Regional Climatological Dispersion

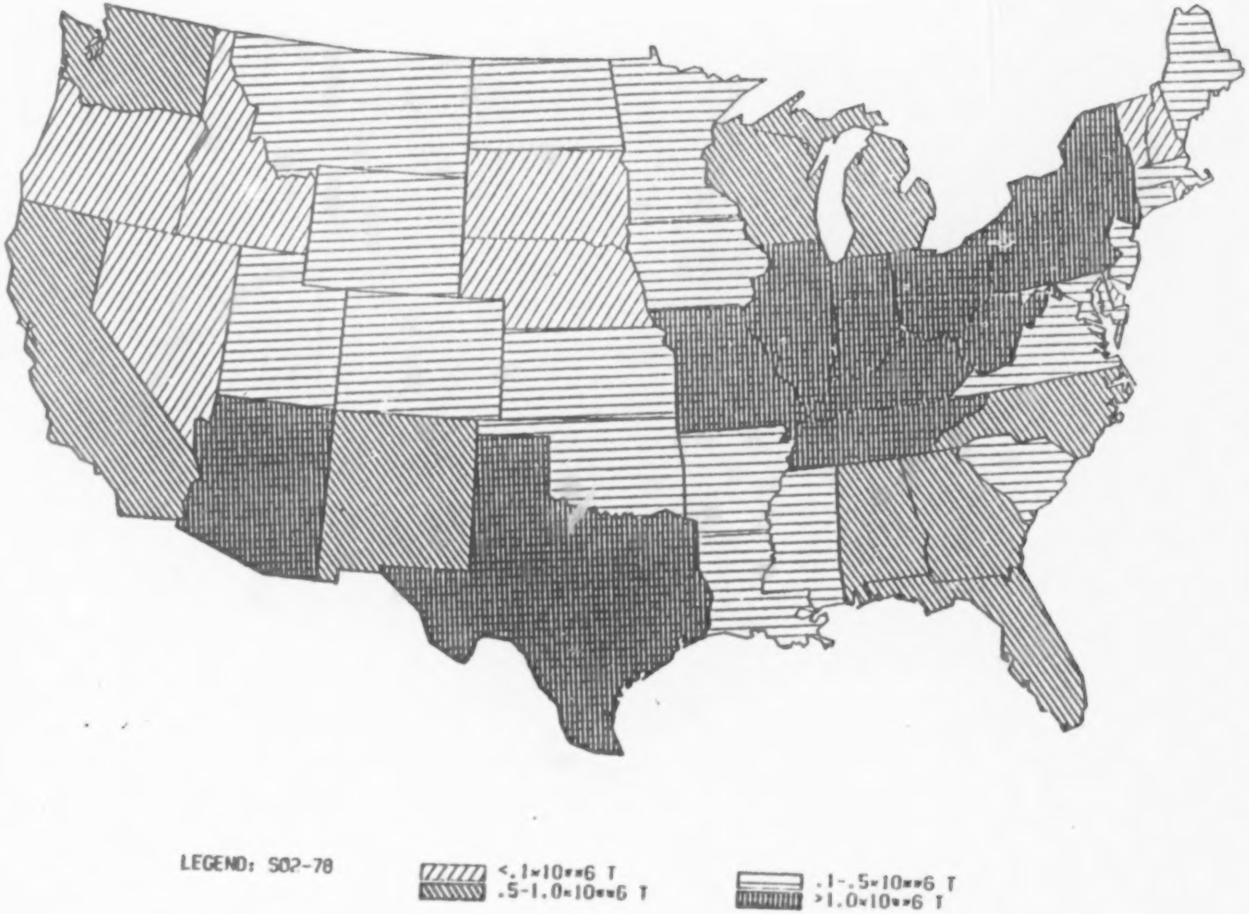
LEGEND: SO2-78
<.1×10**6 T
.5-1.0×10**6 T
.1-.5×10**6 T
>1.0×10**6 T

Figure 2. Total Emissions from all Sectors (tons) - 1978 Baseline

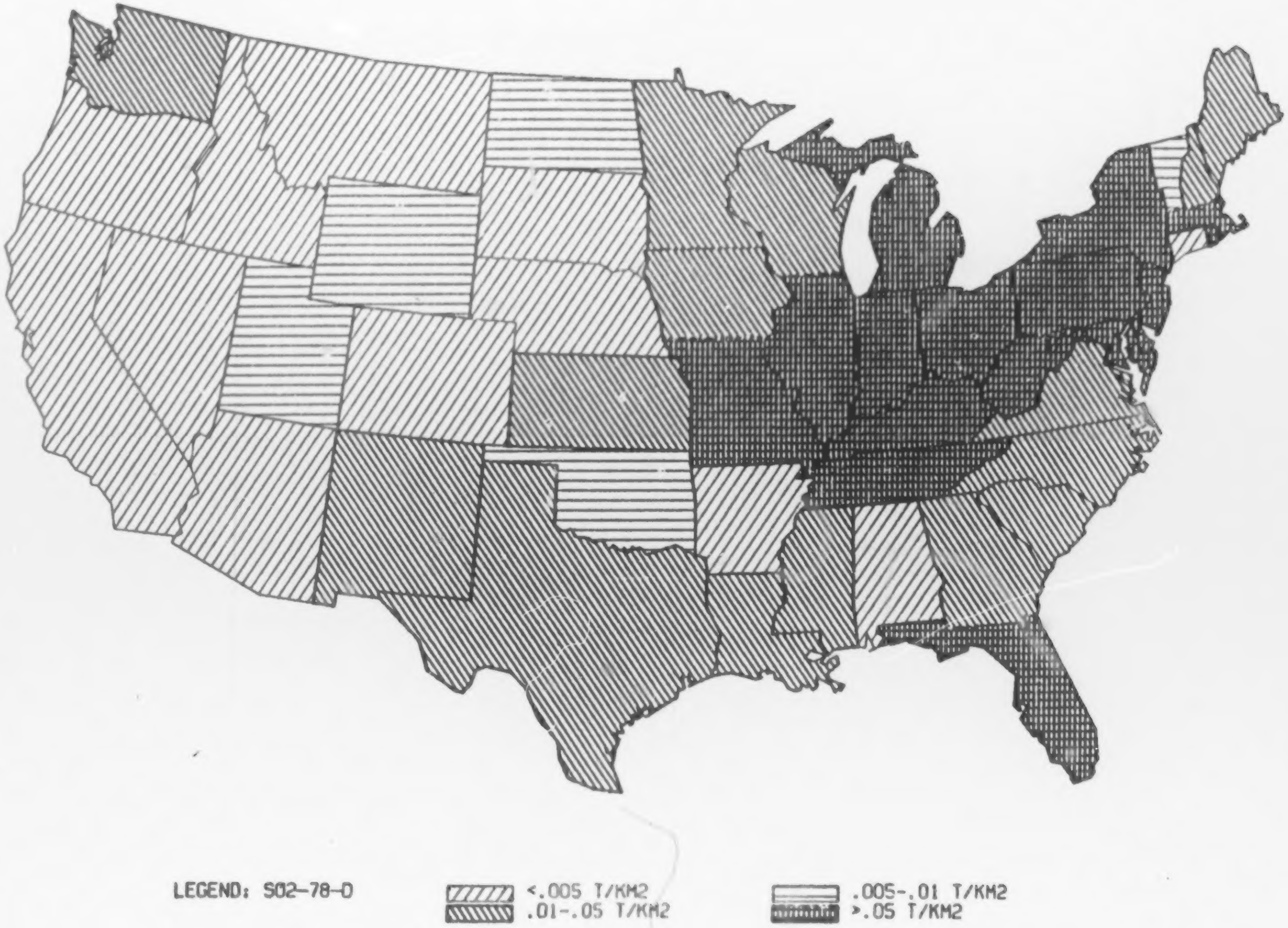
LEGEND: SO2-78-D
▨ <.005 T/KM2
▨ .01-.05 T/KM2
▤ .005-.01 T/KM2
■ >.05 T/KM2

Figure 3. Density of Total SO_2 Emissions from all Sectors (tons/hectare) - 1978 Baseline

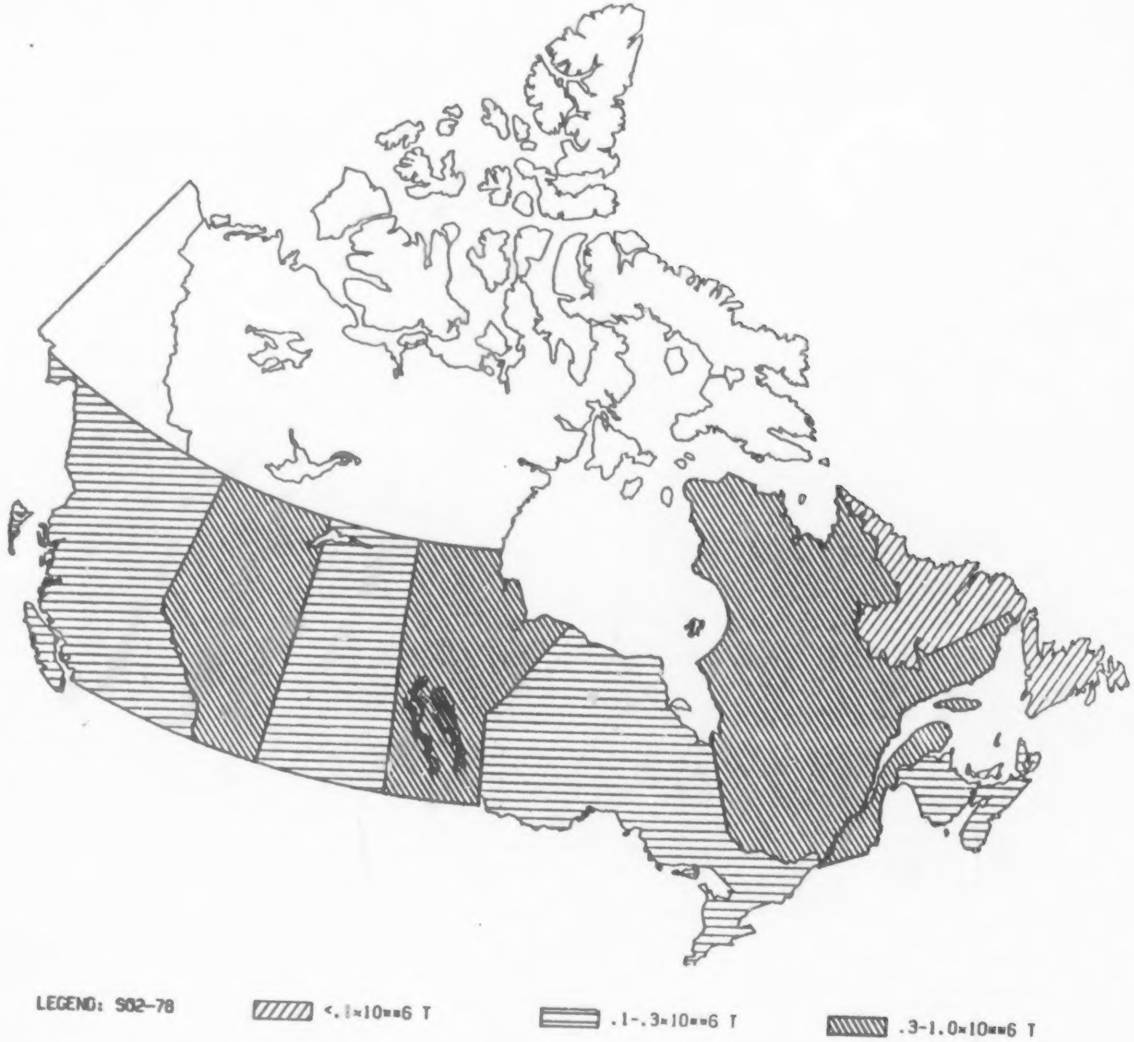
LEGEND: SO2-78
▨ <.1*10**6 T
▤ .1-.3*10**6 T
▧ .3-1.0*10**6 T

Figure 4. Total SO_2 Emissions from all Sectors (tons) - 1978 Baseling

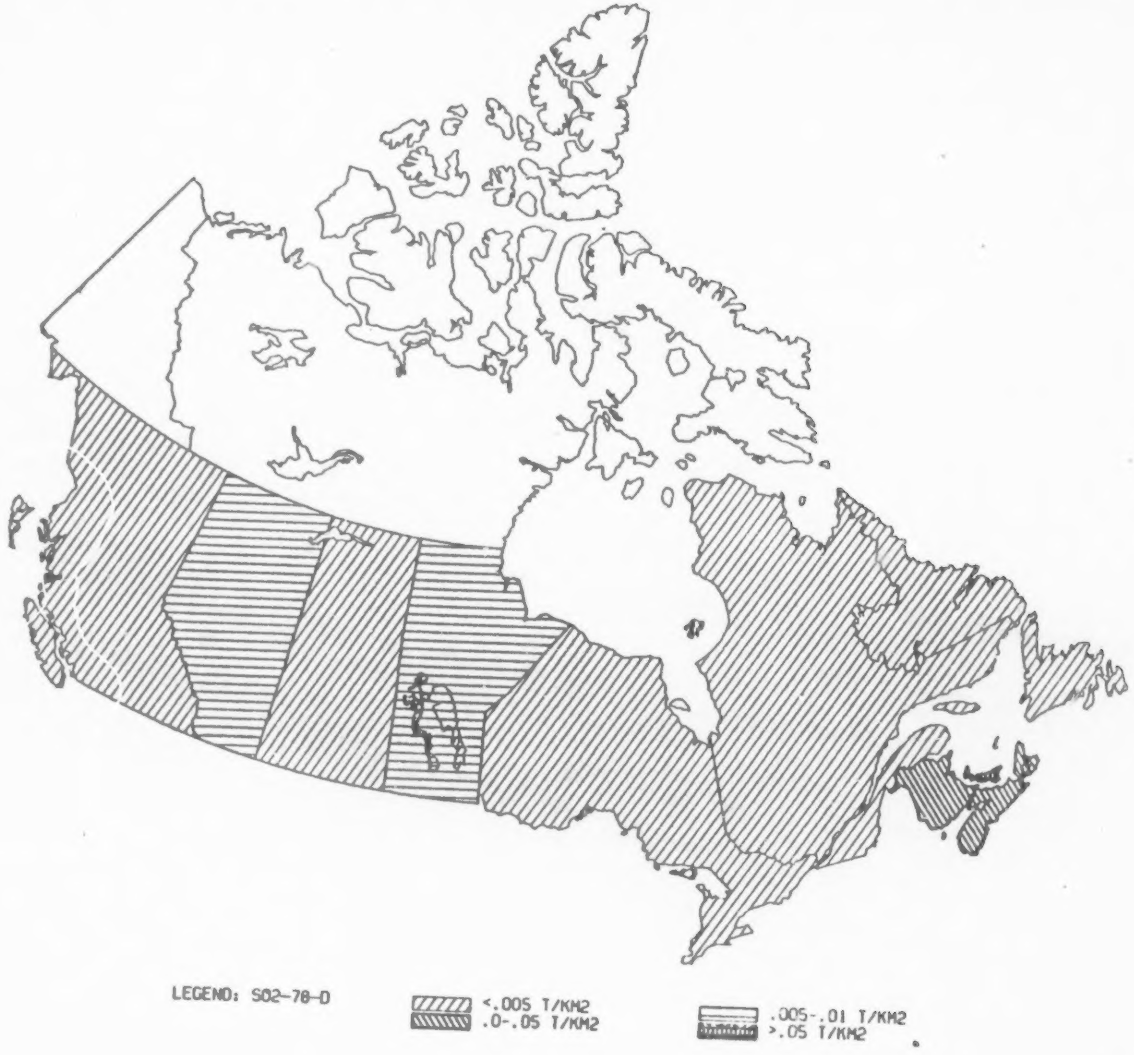
LEGEND: SO2-78-D
▨ <.005 T/KM2
▨ .0-.05 T/KM2
▤ .005-.01 T/KM2
▦ >.05 T/KM2

Figure 5. Density of Total SO_2 Emissions from all Sectors (tons/hectare) 1978 Baseline

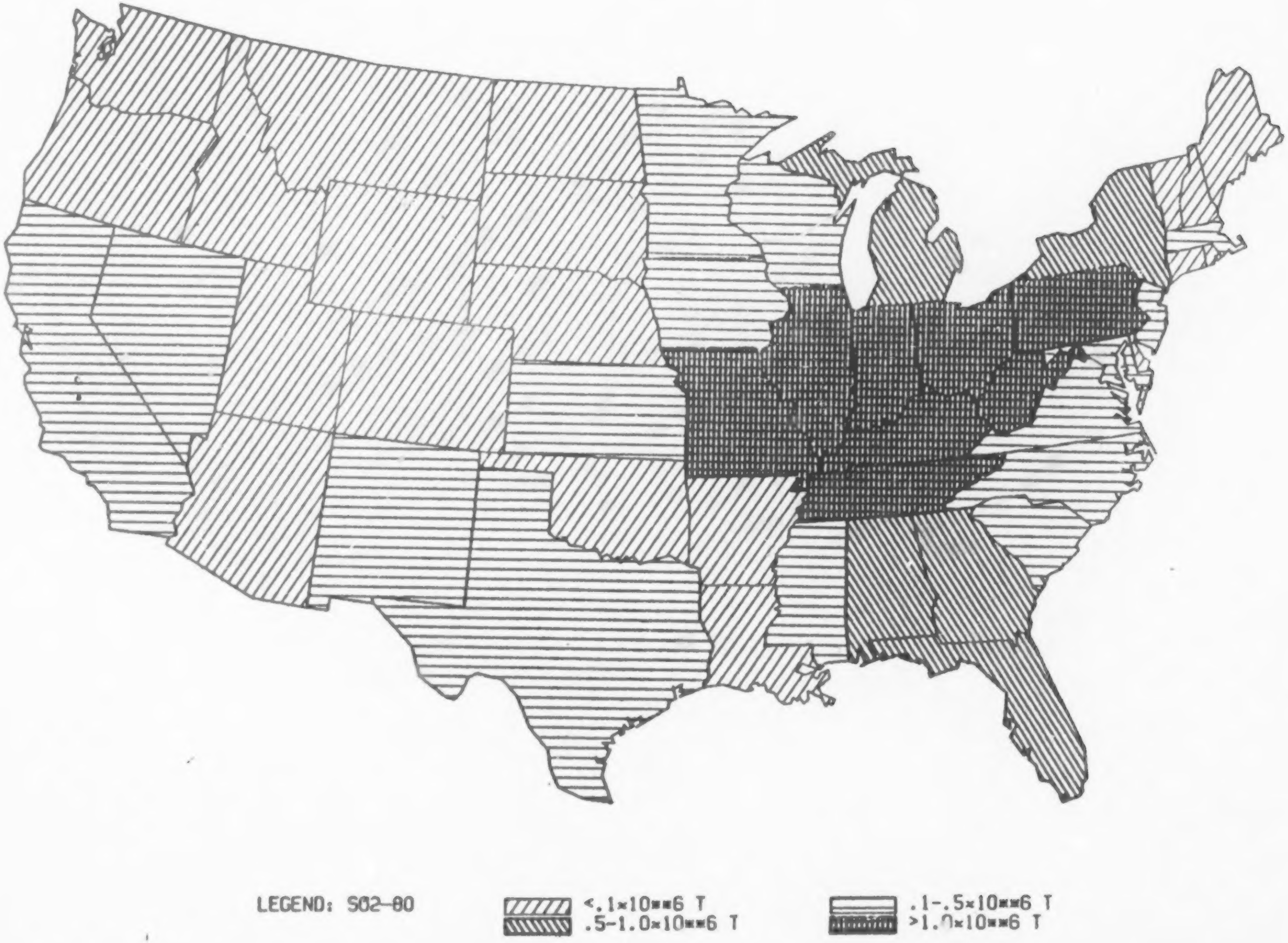
LEGEND: S02-80
▨ <.1×10**6 T
▨ .5-1.0×10**6 T
▤ .1-.5×10**6 T
▦ >1.0×10**6 T

Figure 6. SO_2 Emissions from Electric Utilities (tons) - 1980 Baseling

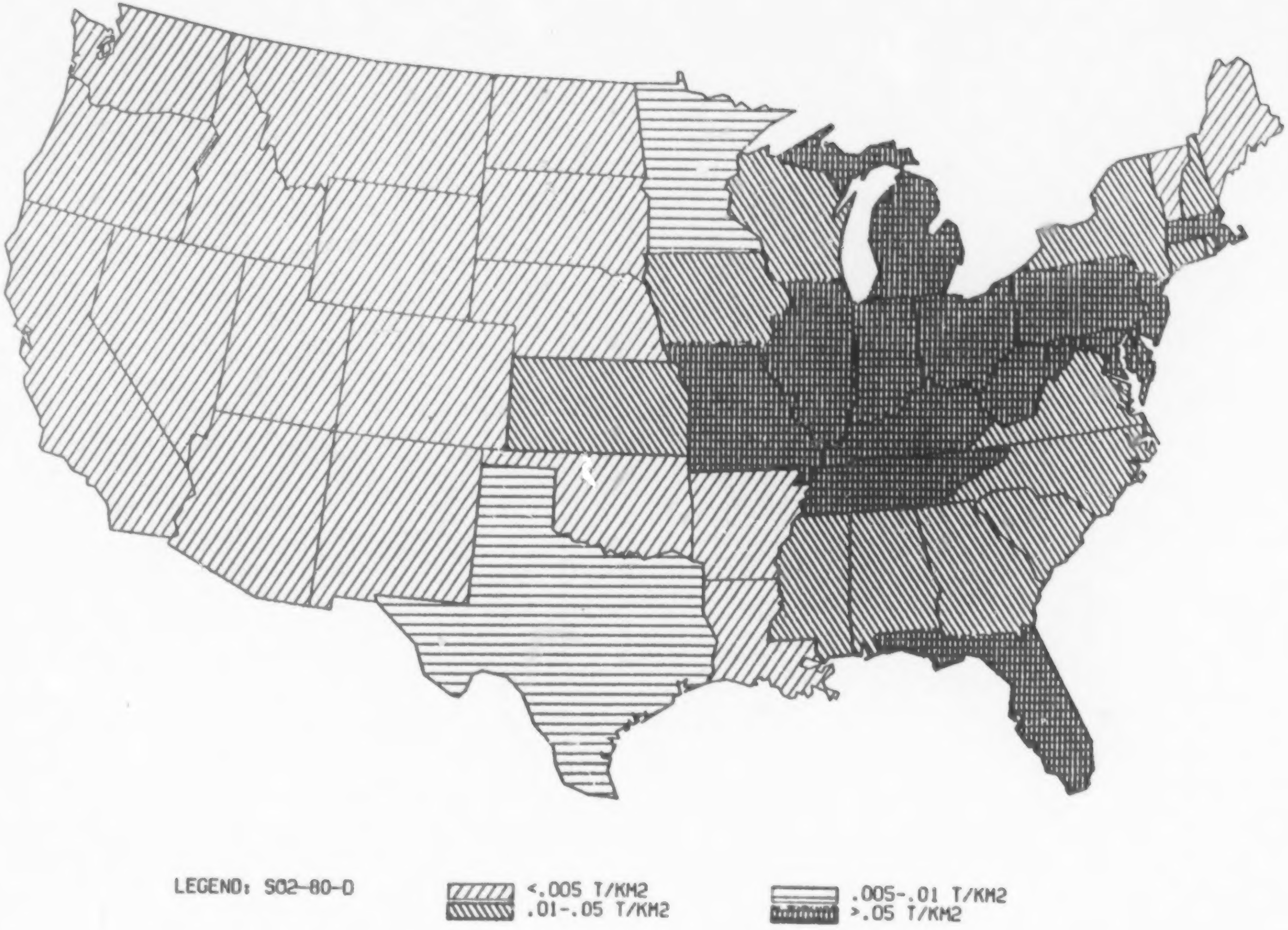
LEGEND: SO2-80-D
<.005 T/KM2
.01-.05 T/KM2
.005-.01 T/KM2
>.05 T/KM2

Figure 7. Density of SO_2 Emissions from Electric Utilities (tons/hectare) - 1980 Baseline

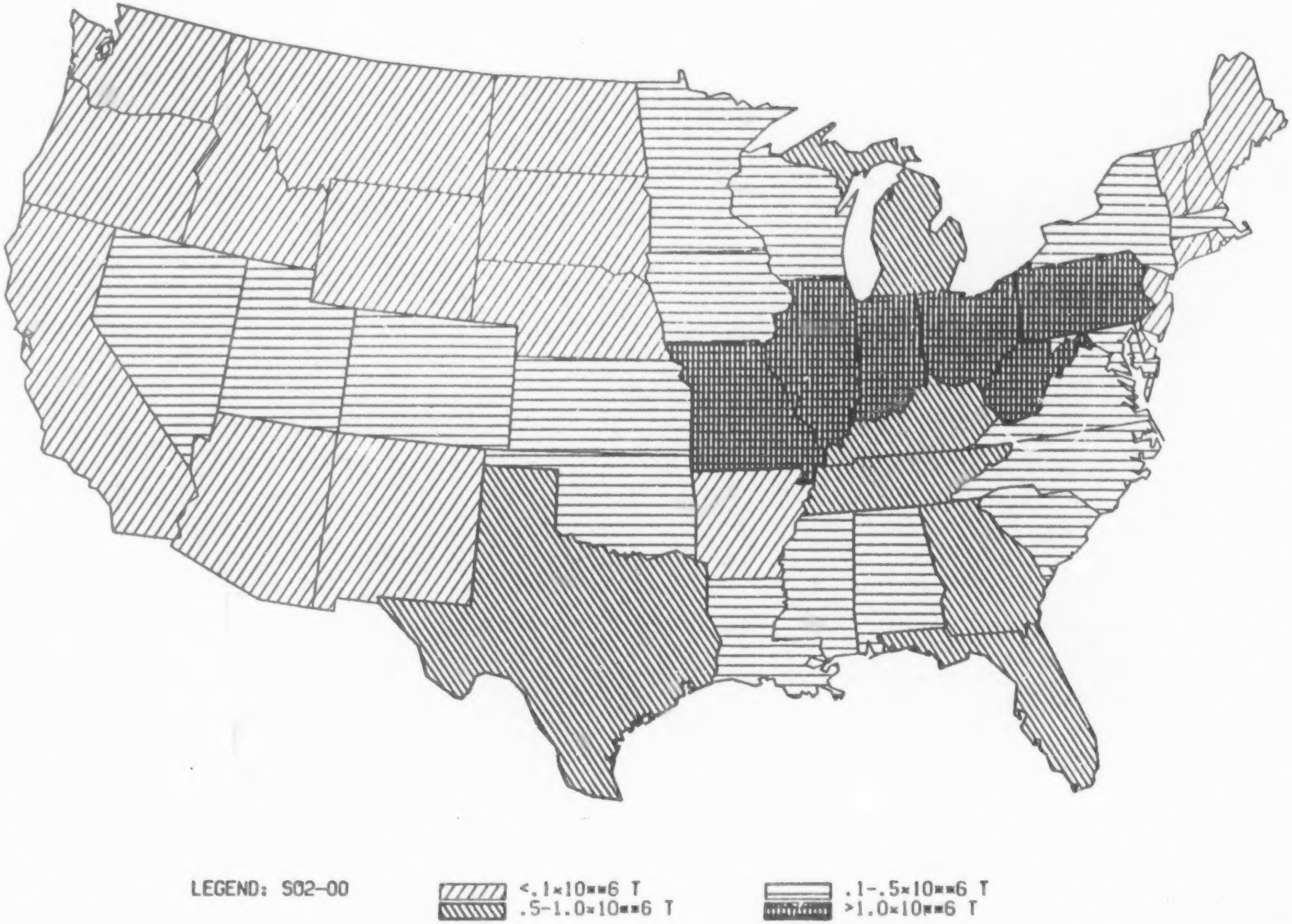
LEGEND: SO2-00
<.1×10**6 T
.5-1.0×10**6 T
.1-.5×10**6 T
>1.0×10**6 T

Figure 8. SO_2 Emissions from Electric Utilities (tons) - 2000 Projection

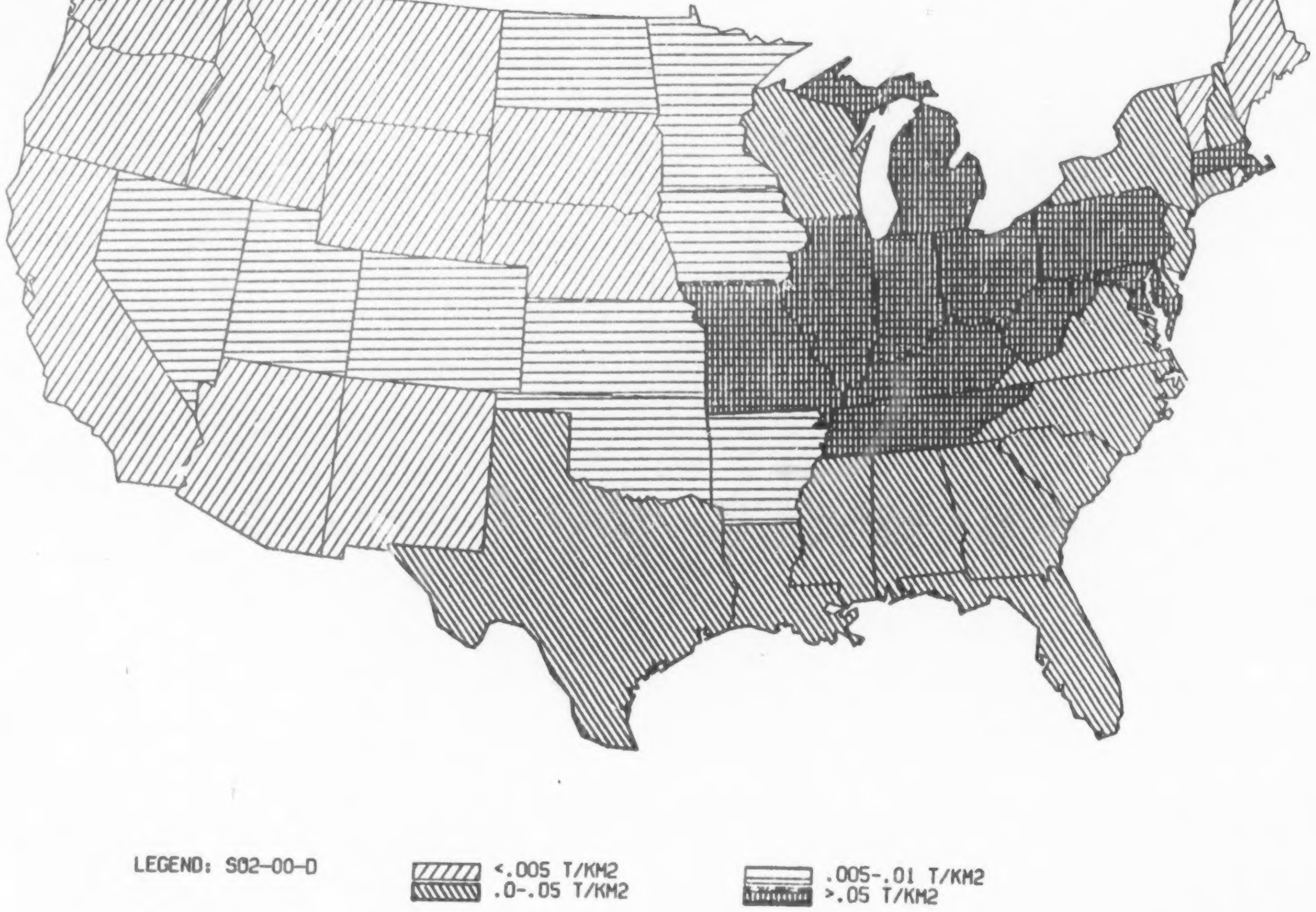
LEGEND: SO2-00-D
<.005 T/KM2
.0-.05 T/KM2
.005-.01 T/KM2
>.05 T/KM2

Figure 9. Density of SO_2 Emissions from Electric Utilities (tons/hectare) - 2000 Projection

Model (RCDM-2) of the University of Illinois. [4]

The SO_2-SO_4 transfer matrix reduces the output of the RCDM-2 model to a set of coefficients, expressing the concentration of sulfates in each state and province caused by a unit of SO_2 emission in each state and province. This 58x58 matrix, showing the influence of each area on itself and on all other areas, was multiplied by the number of units of SO_2 emission in each area to produce an estimate of total sulfate concentration in each area.

The Canadian subprovince-level data generated by the RCDM-2 model were aggregated to the province level to match the level of Canadian population projections. Sudbury, which has unusually high SO_2 emissions, was maintained as a special area. Emissions data were obtained by state and subprovince. Again, the Canadian data were further aggregated to the province level to match the level of population projections.

Because few observations are available for comparing model predictions with observed concentrations, no definite conclusions can be drawn as to the accuracy of the RCDM model. The problem is exacerbated because this model is designed to predict long-term average concentrations, and there are no useful observations of the appropriate term -- about 10 years. Figure 10 compares model results with 1978 observed annual average sulfate concentrations from the EPRI-SURE network (Work Group 2, 1981). Readers should keep in mind that

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[4] RCDM-2 is an adaption of a model developed by Fay and Rosenzweig which, through simplifying assumptions, provides analytical solutions to diffusion equations for sulfur dioxide and sulfate concentrations (Fay and Rosenzweig, 1979). This model is currently under evaluation by the U.S./Canadian Work Group which is comparing results of several different meteorological models (Niemann, 1981).

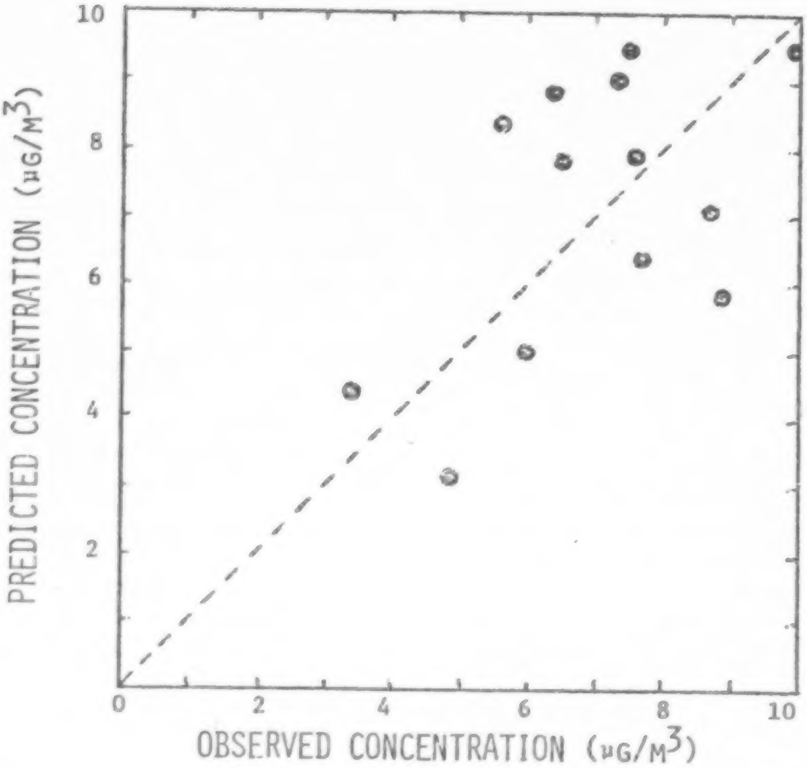


Figure 10. Comparison of RCDM-predicted annual average sulfate concentration with 1978 observations.

an attempt to predict an annual average with a 10-year average model must necessarily be less precise than a prediction based on meteorology solely from the year for which the comparison is being made. The RCDM model underestimates the 1978 data with a mean logarithmic bias of only 0.01 and standard deviation of 0.27. These results are the most accurate and precise available from among the models tested by Work Group 2 at the time of this writing. Given the fact that the RCDM model is based on 10 year averages, its true precision could be better than that indicated by comparisons with annual average observations.

Sulfate concentration estimates are shown in Figures 11, 12, and 13. The concentration distribution for the 30% rollback scenario is identical to the baseline but with a 30% reduced magnitude.

5.2.3 Population Distribution

Data detailing the distribution of population in the United States by state were obtained from the U.S. Bureau of the Census as discussed below. Demographic details (i.e. age, race, sex) were not included.

The 1980 scenario analysis used the recently-reported final first census count reflecting the current distribution of total U.S. population (Figure 14) (U.S. Department of Commerce, 1981). The 2000 scenario analysis used the mo recent projections available from the U.S. Bureau of the Census (U.S. Department of Commerce, 1979), modified to reflect the actual population distribution as identified by the 1980 enumeration (Figure 15).

U.S. Bureau of the Census projection series II-B was chosen as most appropriate for this analysis (U.S. Department of Commerce 1975, 1977). This

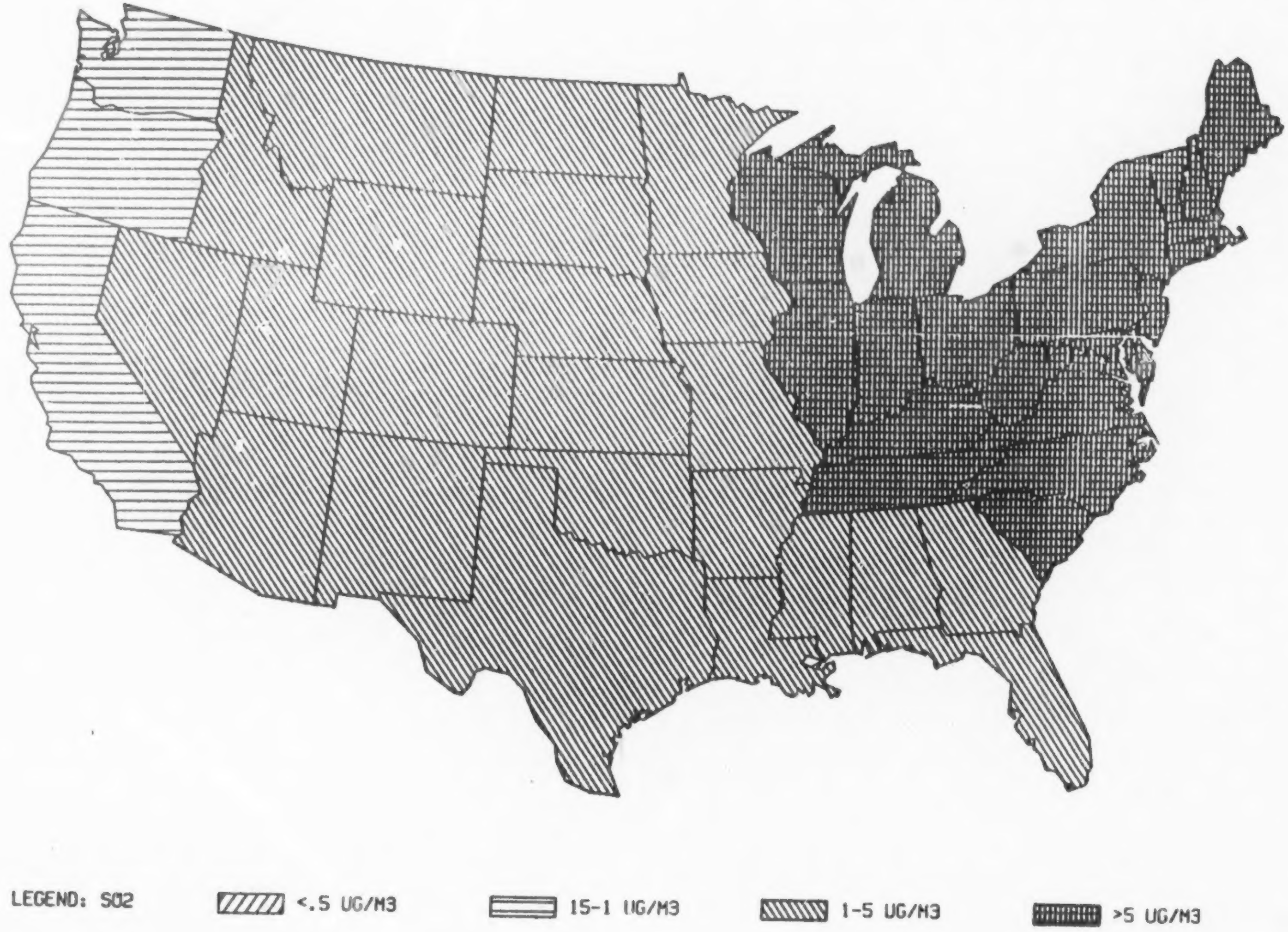

LEGEND: SO2
▨ <.5 UG/M3
▤ 15-1 UG/M3
▧ 1-5 UG/M3
▦ >5 UG/M3

Figure 11. Sulfate Concentration Produced by Total SO_2 Emissions from all Sectors ($\mu g/m^3$) - 1978 Baseline

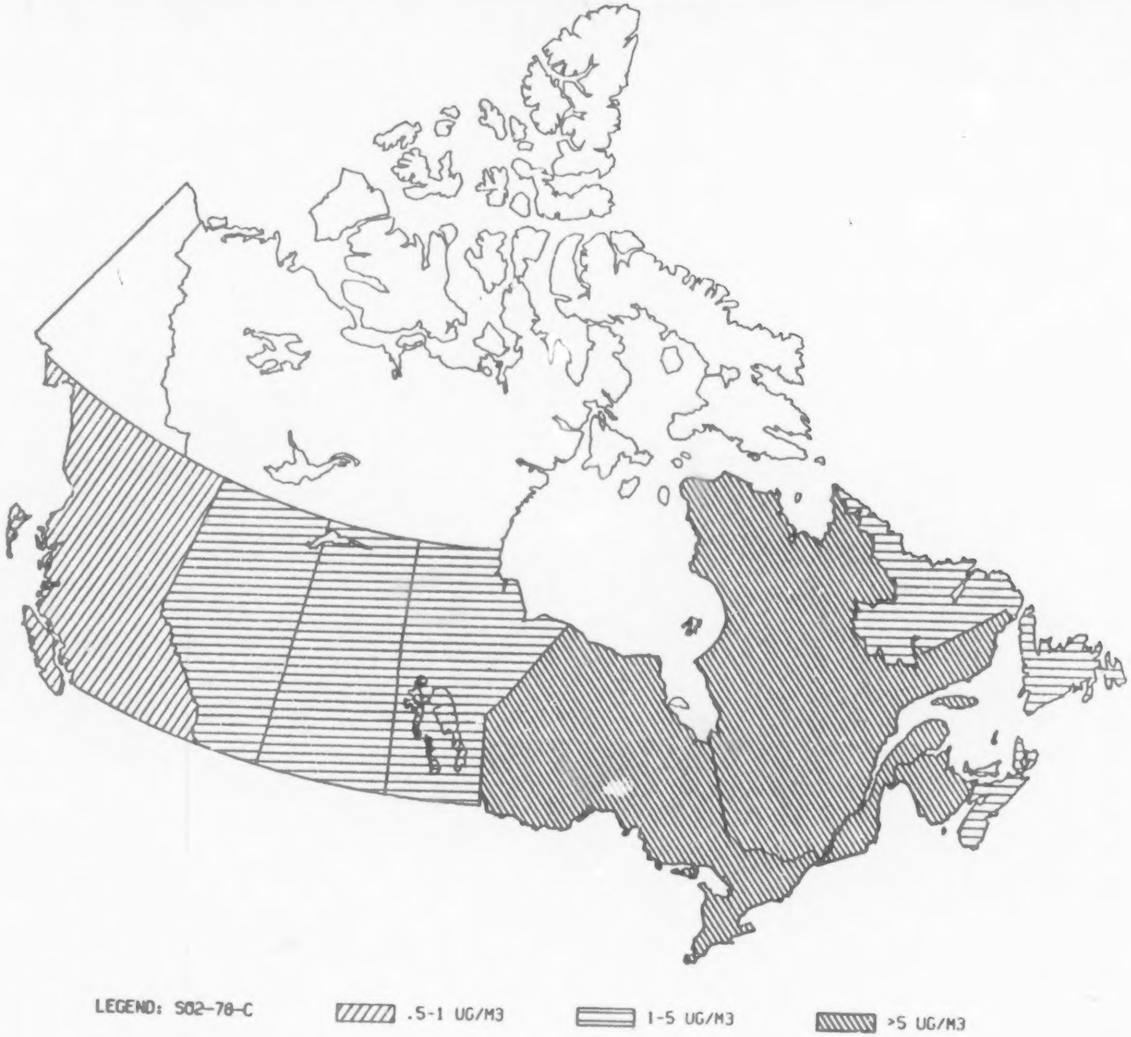

LEGEND: SO2-78-C
▨ .5-1 UG/M3
▤ 1-5 UG/M3
▧ >5 UG/M3

Figure 12. Sulfate Concentration Produced by Total SO_2 Emissions from all Sectors ($\mu g/m^3$) - 1978 Baseline

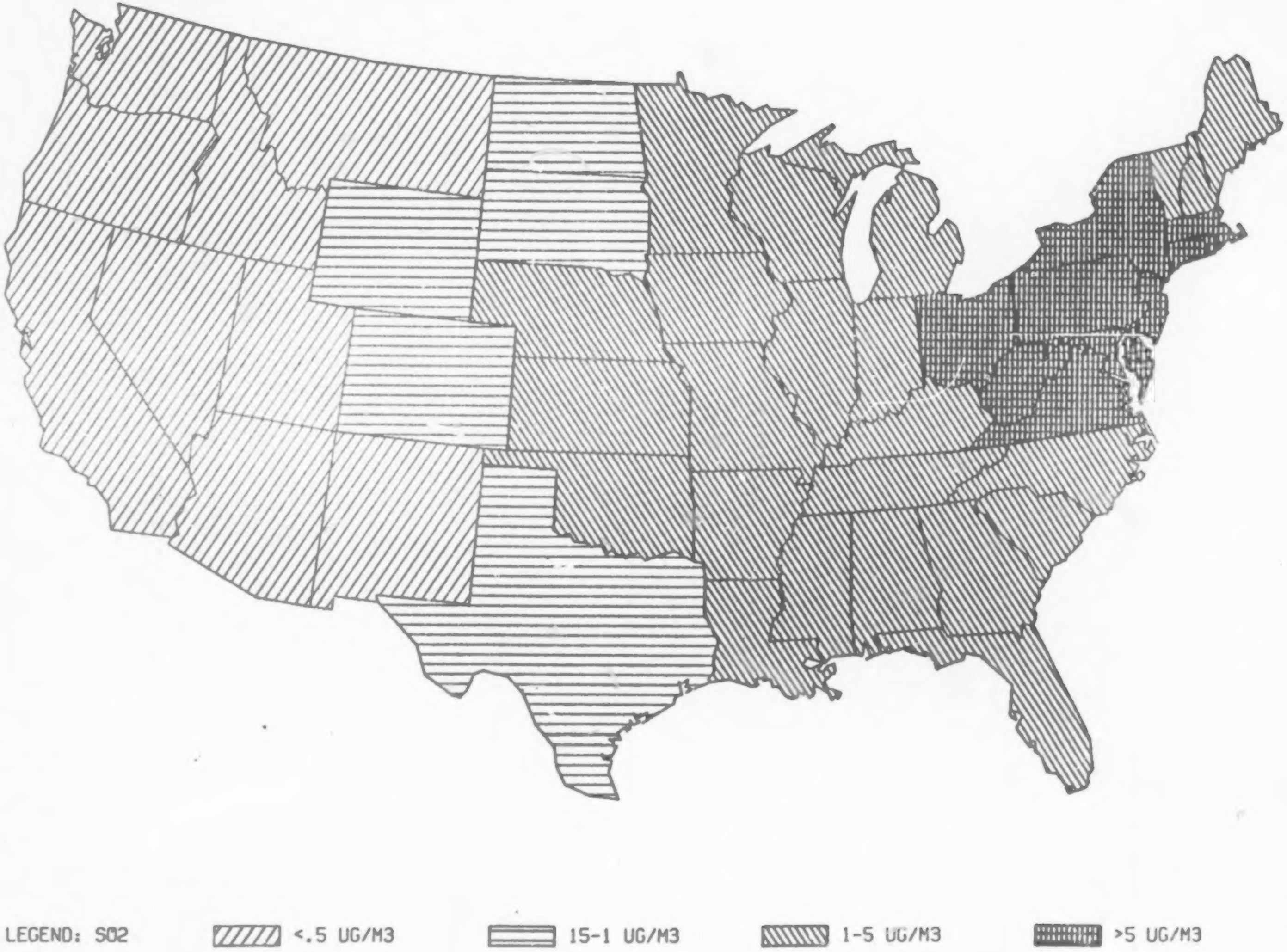
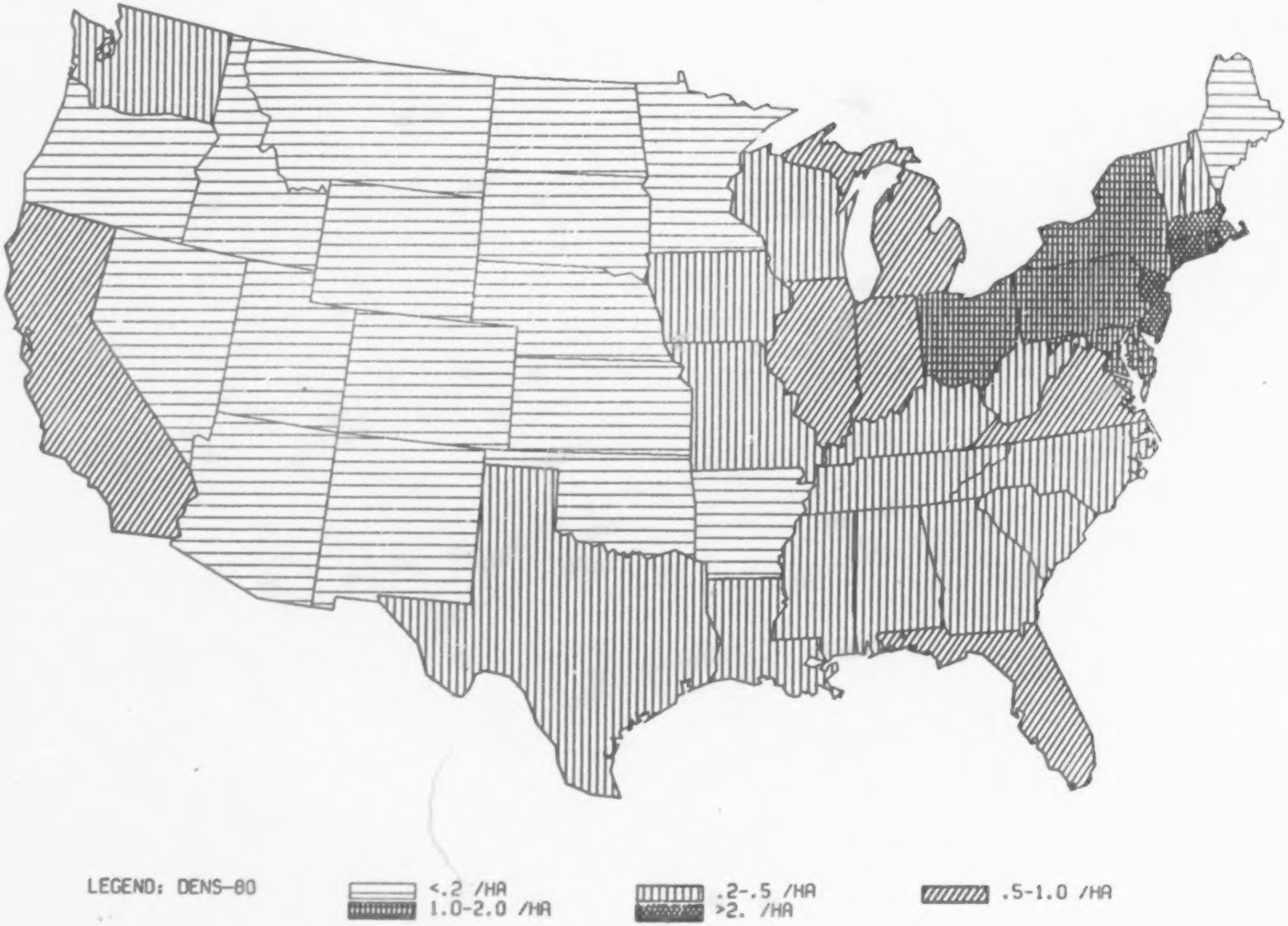

LEGEND: SO2
▨ <.5 UG/M3
▤ 15-1 UG/M3
▧ 1-5 UG/M3
▦ >5 UG/M3

Figure 13. Sulfate Concentration Produced by Total SO_2 Emissions from all Sectors ($\mu g/m^3$) - 2000 Projection



LEGEND: DENS-80
<.2 /HA
1.0-2.0 /HA
.2-.5 /HA
>2. /HA
.5-1.0 /HA

Figure 14. Population Density Distribution - 1980 Baseline

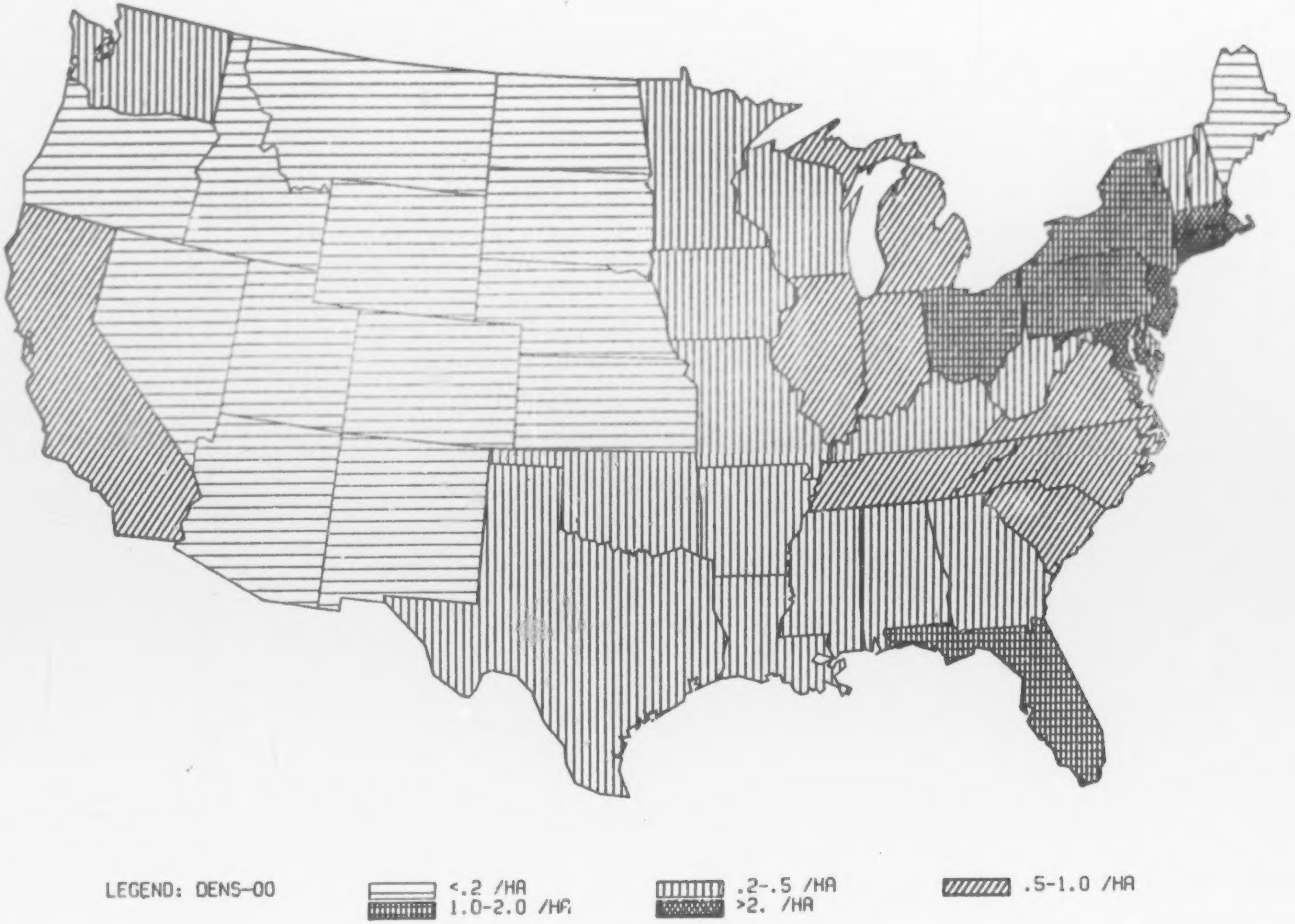

LEGEND: DENS-00
<.2 /HA
1.0-2.0 /HA
.2-.5 /HA
>2. /HA
.5-1.0 /HA

Figure 15. Population Density Distribution - 2000 Projection

series used the estimated July 1, 1976 population as a base year. Other assumptions included: (1) an average of 2.1 lifetime births per woman; (2) life expectancy increases from 69.1 to 71.8 years for males and 77.0 to 81.0 years for females between 1976 and 2050; and (3) a net immigration to the U.S. of 400,000 per year. Series II-B was based on internal migration patterns of the period 1970-75, which characterized the reversal in metropolitan county growth that occurred during that period.

Data detailing the distribution of the population in Canada was obtained from Statistics Canada, Ministry of Industry, Trade, and Commerce (Statistics Canada, 1979) (Figure 16). While Canada recently completed a 1980 census, tabulations were not available for incorporation into this analysis. 1980 projection data were used as surrogates for the unavailable population counts. Projection series No. 5 was selected as most equivalent to U.S. projection series II-B (Basavara, 1981; Wetrogen, 1981). Province-level data, being the closest analogue to state data, was used for comparability with the U.S. data set. No demographic detail was incorporated into the analysis.

5.2.4 Population Exposure

Total population exposure to atmospheric sulfates was estimated as the product of the SO_2 emissions vector, the SO_2-SO_4 transfer matrix, and the population vector for each year or scenario analyzed. Exposure is expressed in person-ugSO_4/m^3.

Exposure to atmospheric sulfates was used as a surrogate for estimating the general health impacts of long-range transport air pollution. We use the probabilistic dose-response function developed by Morgan, et al., (1978),

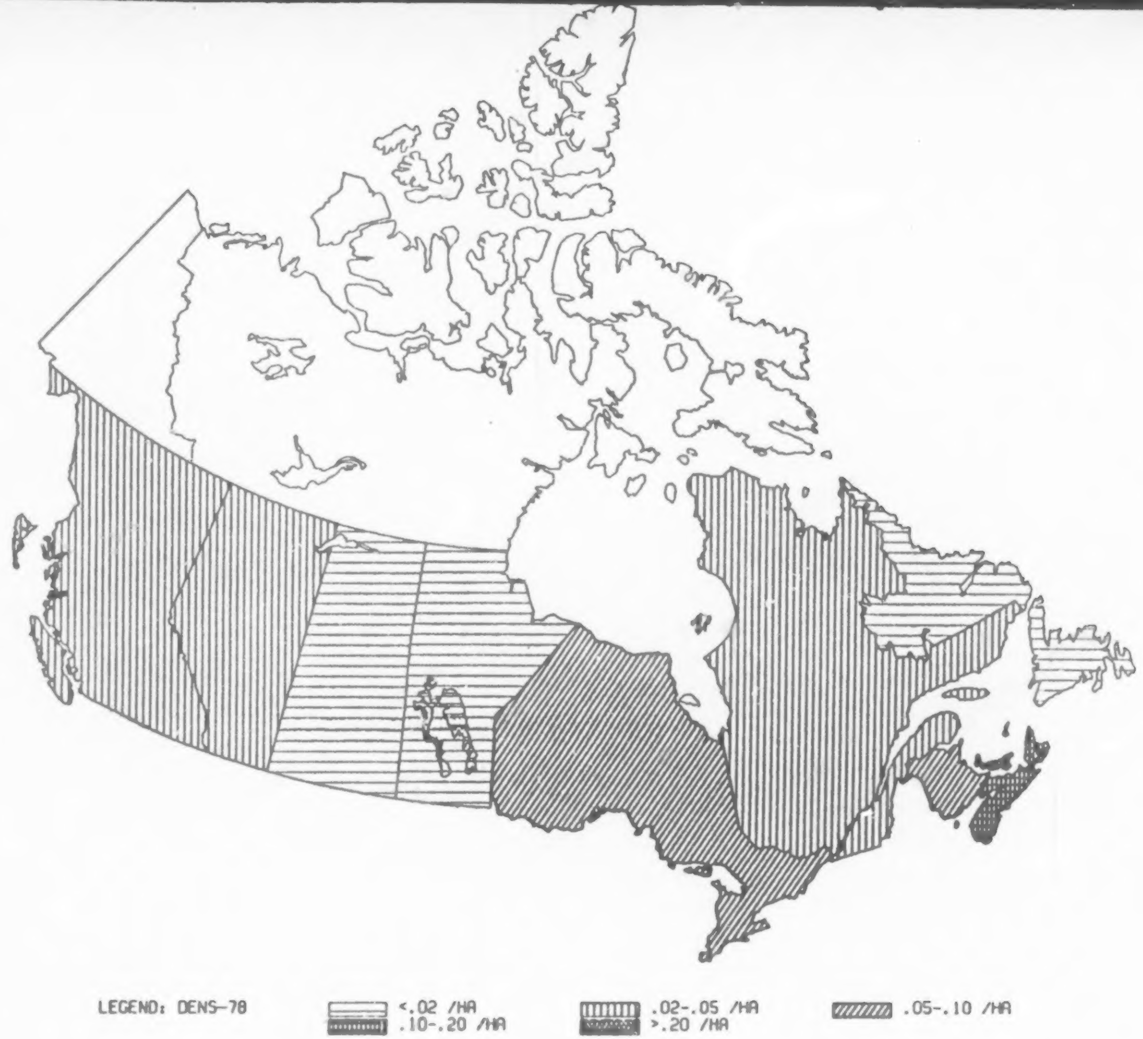

LEGEND: DENS-78
<.02 /HA
.10-.20 /HA
.02-.05 /HA
>.20 /HA
.05-.10 /HA

Figure 16. Population Density Distribution - 1980

shown in Figure 1. It assumes that excess mortality is a linear non-threshold function of sulfate exposure. The slope of the function is expressed as a probability distribution having an expected value of about 3.7 excess deaths per 10^5 person-ug/m^3 and a 90% confidence interval of 0.-11 excess deaths per 10^5 person-ug/m^3.

This method uses atmospheric sulfates as a surrogate or indicator for estimating overall health impacts of the sulfur-particulate mix. Given current knowledge, we believe it to be a reasonable approach. Readers should keep in mind, however, that these health effects estimates rise and fall with SO_2 emissions. This may not adequately capture all of the differences among alternatives or scenarios.

5.3 Results

Detailed results are shown in the Appendices and summarized in Figures 17 and 18. The appendices show distributions of population, SO_2 emissions, sulfate concentration, and excess mortality in each state and province. In reading the appendices, some care must be exercised to distinguish between exposure occuring in an area and exposure created by emissions from that area.

Table 1 summarizes results from all scenarios. Median estimated excess deaths in the base year throughout Canada and the United States are 51,000; there is a 90% confidence range, attributable to uncertainties in the dose-response function, of 0-150,000 excess deaths.

Population is projected to increase by 19% in the interval 1980-2000. If SO_2 emission levels remain unchanged over that interval, excess deaths are projected to increase by 12%. The difference in the rates of change of these

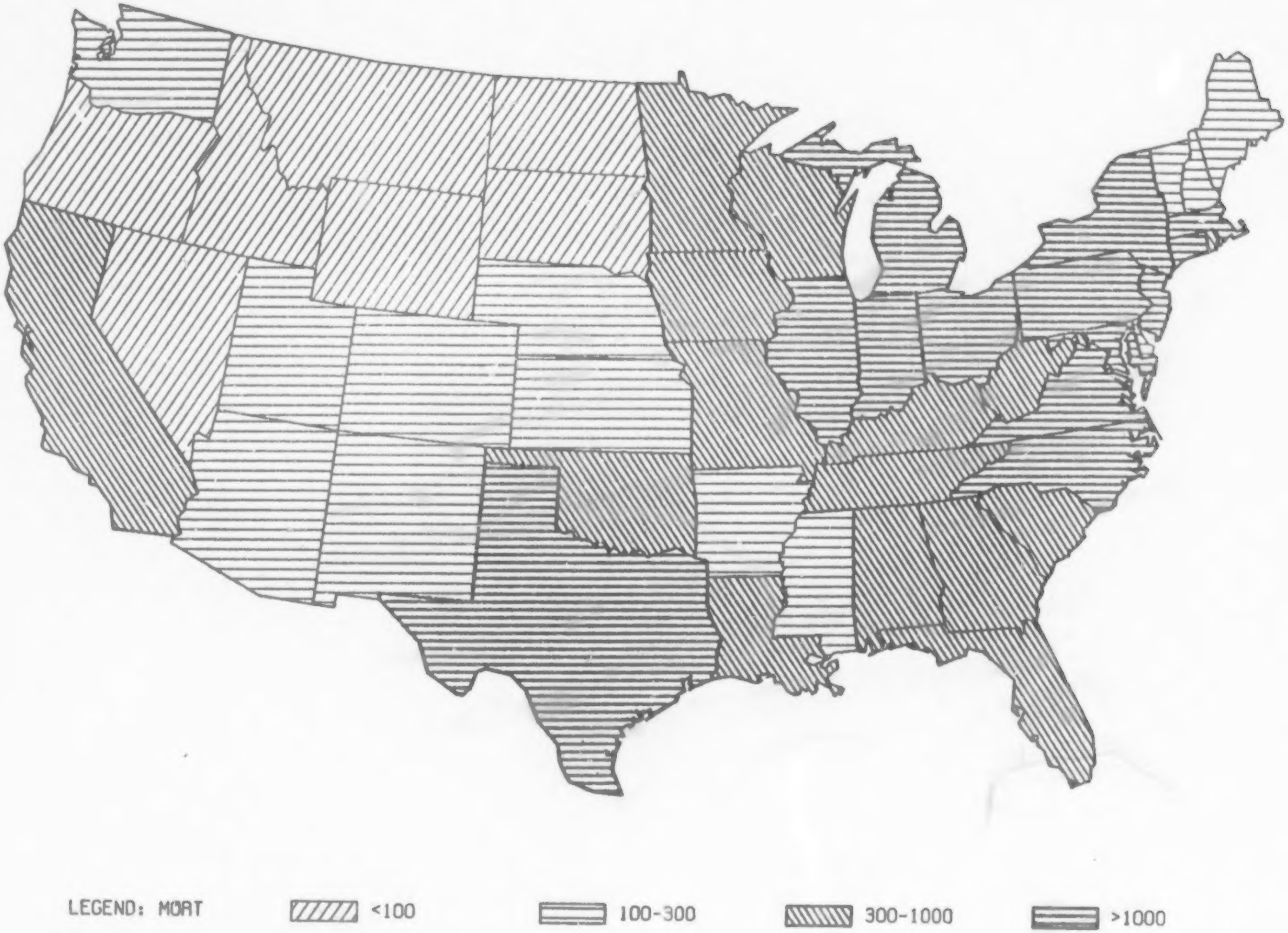

LEGEND: MORT
▨ <100
▤ 100-300
▧ 300-1000
▦ >1000

Figure 17. Excess Deaths Attributable to Total SO_2 Emissions from all Sectors - 1978 Baseline

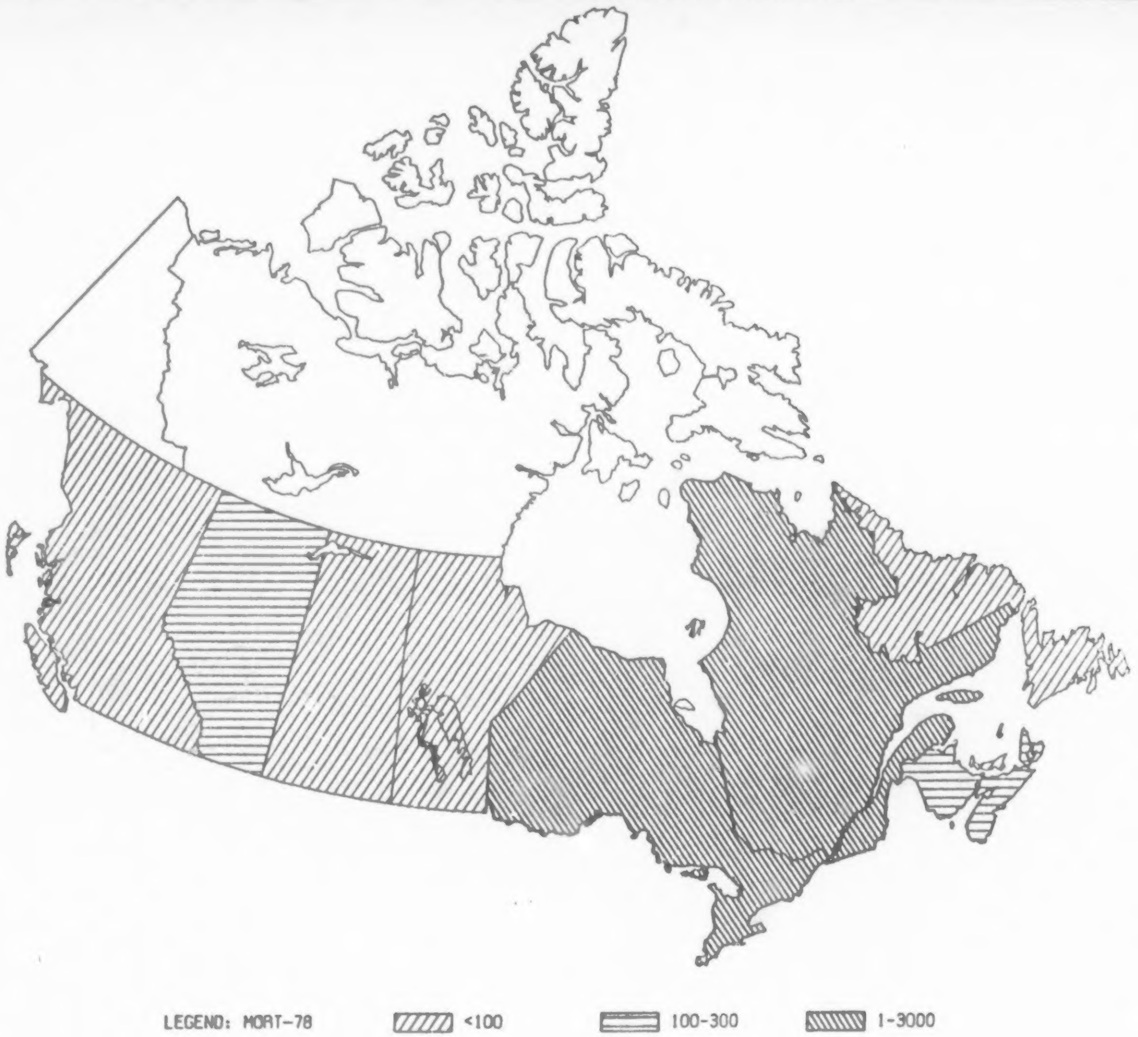

LEGEND: MORT-78
▨ <100
▤ 100-300
▨ 1-3000

Figure 18. Excess Deaths Attributable to Total SO_2 Emissions from all Sectors - 1978 Baseline

two variables is caused by shifts in population distribution. As a result, the mortality rate (deaths per 100,000 population) decreases by 4%. If emission levels decrease uniformly by 30% in all states and provinces, the result is a 22% decrease in estimated total excess deaths and a 36% decrease in mortality rate.

Table 1
Summary of Scenario Results

SCENARIO	Population (millions)	SO_2 Emissions (10^3 tons)	Excess Deaths Total (10^3 deaths)	Range	Rate (Deaths/10^5 Population)
1980 Population, 1978 Total SO_2 Emissions, U.S. and Canada	249	34.	51.	0-150.	20.4
2000 Population, 1978 Total SO_2 Emissions, U.S. and Canada	291	34.	57.	0-170.	19.6
2000 Population, 70% 1978 Total SO_2 Emissions, U.S. and Canada	291	24.	40.	0-120.	13.8

Appendix 3 shows the percentage contribution of each state and province to selected areas in the United States and Canada. These areas represent a broad range of geographic locations, size, and population densities. Note that as a result of long-range transport, some states contribute a smaller percentage to their own sulfate pollution than they do to other states.

5.4 Discussion

These results are best interpreted as mortality rates per 100,000 population. Table 2 shows rates for the emissions scenarios analyzed compared with mortality rates from other sources. Total SO_2 emissions in the United States and Canada are estimated to have increased 1978 mortality rates by about 2%, with a 90% confidence range of 0-7%. Decreases in excess mortality attributable to a 30% emissions rollback by 2000 decrease the relative contribution of total SO_2 to about 1.6%.

The mortality rate attributable to SO_2 emissions from energy production is approximately the same as that from infectious lung diseases. Many of these cases may, in fact, be initiated or exacerbated by exposure to air pollution. The extent of the contribution of pollution to these diseases is unknown.

Table 2
Mortality Rates [5/]

Source	Rate (deaths/100,000 population) Expected	Range
All SO_2 Emissions, U.S. and Canada, 1978	20	0-61
1978 SO_2 Emissions, 2000 population, U.S. and Canada	20	0-61
70% 1978 Emissions, 2000 population, U.S. and Canada	14	0-41
All causes, U.S., 1977[a]	878	
Cardiovascular disease	444	
Cancers	179	
Accidents	48	
Influenza and pneumonia	24	
Bronchitis, emphysema, and asthma	10	
Suicides	13	

[a]Source: Statistical Abstracts of the United States, 1979. Bureau of the Census, U. S. Department of Commerce.

On average, both for the United States and Canada, about 90% of the total health effects caused by emissions in one area are exported to other areas and

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[5/] Detailed emissions projections for 2000 exist only for utilities. If non-utility emissions change proportionally with utility emissions, the total mortality rate attributable to SO_2 emissions is estimated to be 16 excess deaths/100,000 population. If non-utility emissions remain either unchanged or emissions increase in proportion to the increase in population, then excess deaths are estimated to be 17 and 18/100,000 population, respectively. The 90% confidence range on these estimates attributable to uncertainty in the dose-response function is about 0-50.

about 90% of the total health effects within each area are imported from other areas. This multi-state and -province sharing of health impacts from SO2 emissions has obvious potential for large-scale inequities. Since few of the areas analyzed must bear a significant portion of the total health impact burden created by their own SO_2 emissions, and most of the areas receive major sulfate contributions from other sources, the equity question pertains to the ratio between imported and exported health effects. If each area imports exposure roughly in proportion to the amount it exports, then the long-range transport does not create inequities. Inequities arise when areas import relatively large exposure burdens compared to their exports or vice versa. Figures 19 and 20 show the distribution of the ratio of imported to exported sulfate exposure in each area in 1978. In general, the northeastern and western United States and southeastern and western Canadian provinces have a ratio greater than 1.0 (net importers) and the central states and provinces have a ratio less than 1.0 (net exporters). Table 3 shows the percent of the states and provinces with ratios of <0.5, 0.5-2.0, and >2.0. Of course, this ratio is only one variable in the multifaceted equity "equation" and many other considerations should be included in any general evaluation of equity.

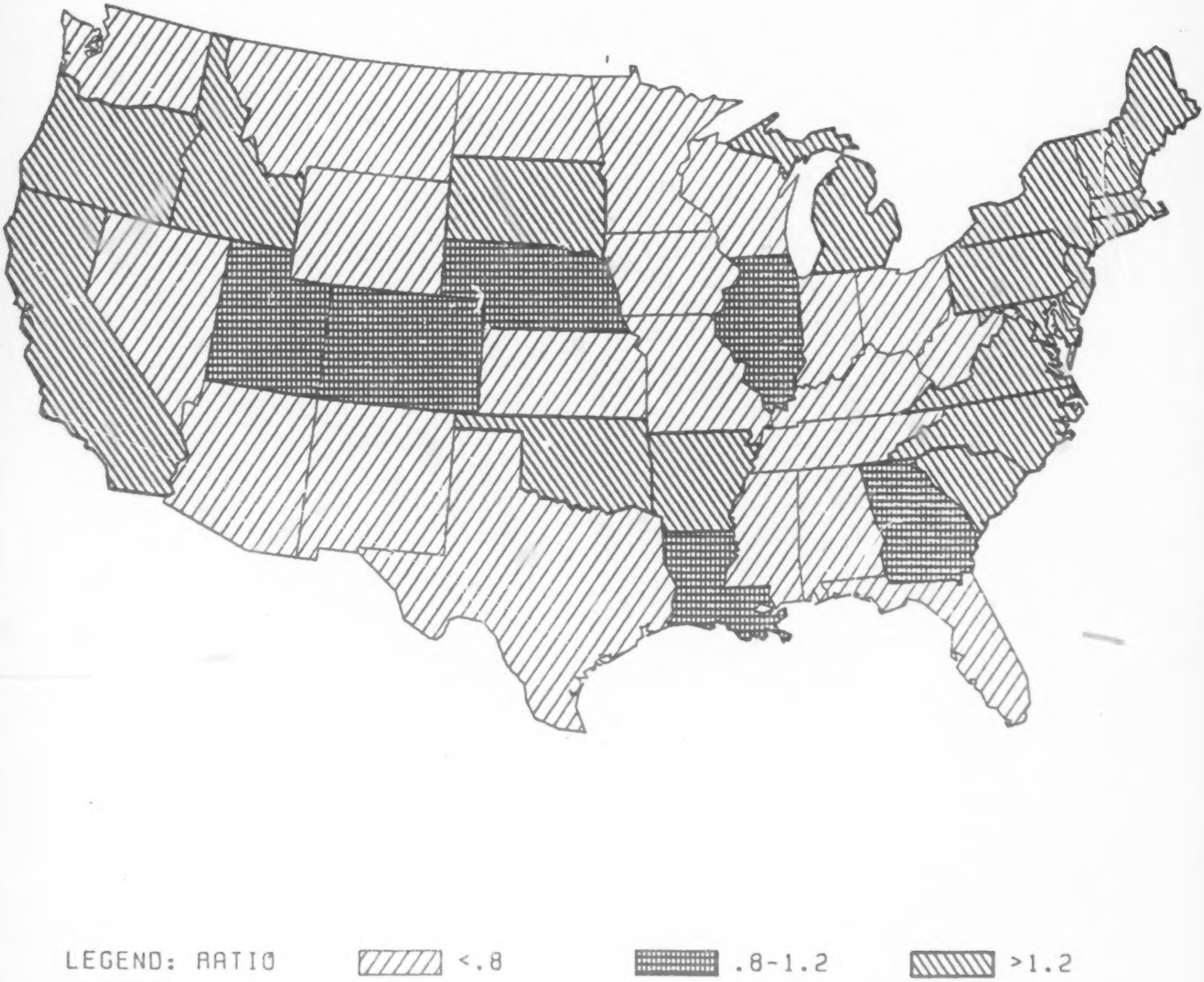
LEGEND: RATIO
<.8
.8-1.2
>1.2

Figure 19. Ratio of Imported/Exported Sulfate Exposures from Total SO_2 Emissions from all Sectors - 1978 Baseline

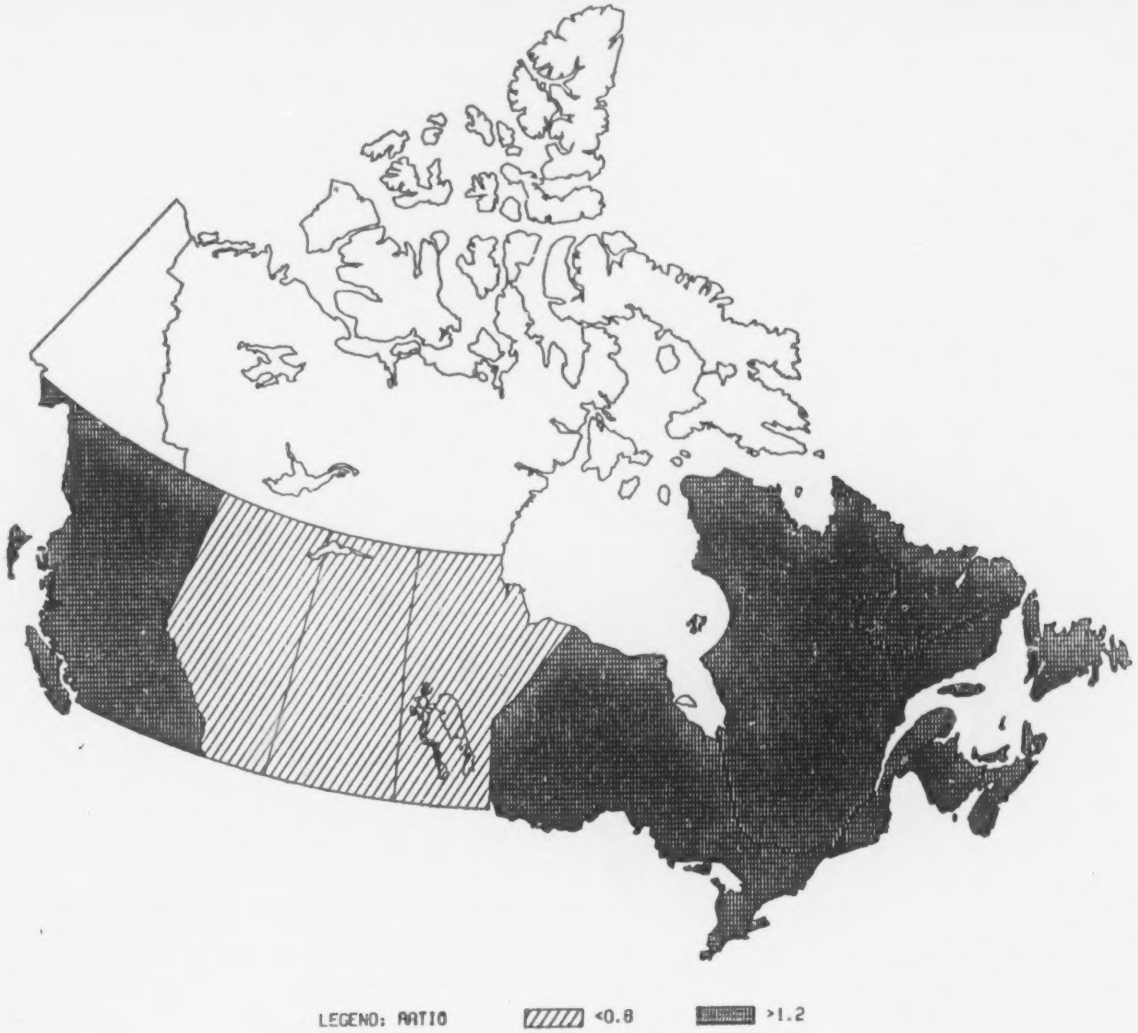
LEGEND: RATIO
▨ <0.8
▩ >1.2

Figure 20. Ratio of Imported/Exported Sulfate Exposures from Total SO_2 Emissions from all Sectors - 1978 Baseline

Table 3
Distribution of Import: Export Ration - 1978

Ratio	% of States and Provinces
<0.5	17
0.5-20	31
>2.0	52

An important international consideration is the relative distribution of health impacts transferred between the United States and Canada. The total exposure exported from the United States to Canada in 1978 is 94 million person-ug/m^3. This represents about 2,000-3,000 excess deaths exported from each country every year, or an imported mortality rate of 14 excess deaths/100,000 population in Canada and 1.0 excess deaths/100,000 population in the United States.

Readers should keep in mind that the health impacts reported here, and the equity considerations generated by health impacts attributable to long-range transport, depend heavily on the assumption of a linear non-threshold dose-response function. Should there be, as some researchers assert, a concentration threshold below which sulfates have no health effects, the distribution of health impacts and equity considerations would differ from those presented here.

APPENDIX JJ-A

The early Lave-Seskin studies have been subjected to considerable criticism based largely on the kinds of difficulties discussed above (Lipfert 1978; Thibodeau, et al., 1980; Smith, 1976). Their 1960 air pollution data set has been criticized for incorporating measurements from 1957, 1958 and 1959, and because air sampling schedules varied among cities. Since the variables used in the Lave-Seskin analysis can only be construed as an index of integrated long-term pollution exposure, the fact that their data were from different years is a minor issue. In any event, Lave and his co-workers have replicated their findings using 1969 and 1974 pollution and health data, so this issue is of little more than historical interest. Questions of the validity of the older measurements also seem of little current concern in light of the replication with later data. A minor but continuing problem is that the later model was developed to fit the 1960 data. An original analysis using the more recent data might lead to a slightly different model formulation.

Smith (1976) found that many possible formulations of the Lave-Seskin results would have been invalid; he reports, however, that all important assumptions required by statistical theory hold for the formulations Lave and Seskin actually used. The one assumption that does not hold was that of normally distributed errors with constant variance. This failure is expected a priori and does not have major substantive implications. The regression coefficients are still valid, but caution is required when making probabilistic statements, such as statements about confidence intervals of pollution regression coefficients or tests of hypotheses. Error rates may be slightly different from those stated.

The variables Lave and Seskin used are highly collinear. For a model selected in advance (as in the replications of the Lave-Seskin analysis) multicollinearity simply produces increased uncertainty in the regression coefficients. Under controlled experimental conditions, researchers try to eliminate multicollinearity, but observational work does not have this advantage. Replication of the Lave and Seskin findings, however, is powerful evidence of substance in the model.

Concern has been expressed that if health damage is caused by exposure over many years, then observed effects should reflect earlier (and much higher) air pollution levels. If reductions in concentration leave the ranking of cities unchanged, the association between pollution and mortality would remain valid, but the size of the derived coefficient would be overstated. The change in average sulfate coefficients over 14 years has not been large relative to uncertainty in the data, but the coefficients do decrease. This is expected during a period of declining pollution levels given a lag time in exposure-effect, although there may be other explanations.

Observational studies using multi-city and single-city methods cover a wide range of population sizes. Single-city studies have examined Buffalo, Nashville and Pittsburgh among others, using TSP as a pollution index rather than sulfates. The Lave-Seskin studies examine all Standard Metropolitan Statistical Areas in the United States with usable air pollution information. Recently completed work by Bozzo, et al., (1979) and Mendelsohn and Orcutt (1979) examine all counties in the United States. This powerful expansion of sample size has yielded results consistent with the earlier studies.

The central issue in evaluating the Lave-Seskin results is that they have been thoroughly reviewed and have withstood criticism for ten years. While it is possible that a key variable was overlooked, none has been demonstrated to date. Addition of factors such as smoking and water hardness has not significantly affected the importance of air pollution in the regressions, although the coefficients for sulfates can change considerably. As time passes and no such variables are found, confidence increases that the findings can be given even more credence.

On the other hand, uncertainty in the magnitude of the resulting coefficients is likely to be much higher than a standard t-statistic would indicate, because there is also uncertainty about the specific agent; identifications of sulfate as the probably causative agent rely largely on animal and clinical studies.

APPENDIX 1

Population, Emissions, and Sulfate Concentration - 1978 Baseline
United States

State	Population (millions)	SOx Emissions (1000 tons)	Sulfate Concentration (ug/m^3)
Ohio	10.80	3100	9.6
Illinois	11.42	1489	6.2
Pennsylvania	11.87	2067	10.5
Indiana	5.49	2065	7.8
Kentucky	3.66	1277	7.0
Michigan	9.26	988	7.6
Tennessee	4.59	1116	5.8
Missouri	4.92	1296	4.3
West Virginia	1.95	1140	9.2
New York	17.56	1109	10.0
Alabama	3.89	694	3.8
Wisconsin	4.71	672	5.3
Iowa	2.91	357	4.2
Minnesota	4.08	400	3.3
Virginia	5.35	387	8.9
North Carolina	5.87	573	7.0
Florida	9.74	927	2.6
Georgia	5.46	641	4.5
South Carolina	3.12	287	5.5
Maryland	4.22	318	9.6
Delaware	.60	87	9.5
New Jersey	7.36	367	9.7
District of Columbia	.64	40	9.6
Arkansas	2.29	111	3.4
Louisiana	4.20	258	2.2
Mississippi	2.52	199	3.1
Massachusetts	5.74	348	9.2
Connecticut	3.11	142	9.4
Rhode Island	.95	30	9.0
Maine	.62	105	6.8
Vermont	.69	15	8.7
New Hampshire	1.57	84	8.3
North Dakota	1.12	138	2.2
South Dakota	.51	26	2.5
Nebraska	.92	94	2.8
Kansas	2.36	327	3.1
Oklahoma	3.03	158	2.8
Texas	14.23	1669	2.0
Montana	.79	148	1.6
Wyoming	.47	218	2.1
Colorado	2.89	291	2.6
New Mexico	1.30	696	2.3
Idaho	.94	37	1.3
Utah	1.46	148	1.9
Arizona	2.72	2335	1.9
Washington	4.13	518	1.0
Oregon	2.63	52	.9
Nevada	.80	77	1.3
California	23.67	794	.9

APPENDIX 1

Population, Emissions, and Sulfate Concentration - 1978 Baseline
Canada

State	Population (millions)	SOx Emissions (1000 tons)	Sulfate Concentration (ug/m^3)
Manitoba	1.04	442.	1.6
Ontario	8.47	240.	6.7
Sudbury	.10	1114.	7.4
Quebec	6.33	368.	5.3
New Brunswick	.71	221.	5.7
Nova Scotia and PEI	1.12	254.	4.8
Newfoundland & Labrador	.58	61.	2.4
Saskatchewan & Alberta	3.00	642.	1.5
British Columbia	2.58	229.	.8
Totals	249.06	33984.	

APPENDIX 2

Health Impacts - 1978 Baseline
United States

State	Sulfate Exposure (million P-ug/m^3)	Excess Mortality (Deaths)		Mortality Rate (Deaths /100000 P)
		Median	90% Range	
Ohio	103	3820	0- 11356	35.4
Illinois	70	2602	0- 7735	22.8
Pennsylvania	125	4619	0- 13733	38.9
Indiana	43	1583	0- 4707	28.8
Kentucky	26	945	0- 2810	25.8
Michigan	71	2614	0- 7771	28.2
Tennessee	27	984	0- 2925	21.4
Missouri	21	784	0- 2330	15.9
West Virginia	18	666	0- 1980	34.2
New York	176	6500	0- 19324	37.0
Alabama	15	550	0- 1635	14.1
Wisconsin	25	926	0- 2753	19.7
Iowa	12	450	0- 1337	15.4
Minnesota	13	492	0- 1463	12.1
Virginia	48	1758	0- 5228	32.9
North Carolina	41	1512	0- 4494	25.7
Florida	25	928	0- 2760	9.5
Georgia	25	918	0- 2730	16.8
South Carolina	17	633	0- 1883	20.3
Maryland	40	1497	0- 4451	35.5
Delaware	6	208	0- 618	35.0
New Jersey	72	2655	0- 7893	36.1
District of Columbia	6	227	0- 675	35.6
Arkansas	8	287	0- 854	12.6
Louisiana	9	349	0- 1037	8.3
Mississippi	8	285	0- 847	11.3
Massachusetts	53	1945	0- 5782	33.9
Connecticut	29	1082	0- 3217	34.8
Rhode Island	9	316	0- 939	33.3
Maine	8	283	0- 841	25.1
Vermont	4	165	0- 492	32.4
New Hampshire	8	284	0- 845	30.9
North Dakota	1	54	0- 161	8.3
South Dakota	2	65	0- 193	9.4
Nebraska	4	163	0- 483	10.4
Kansas	7	267	0- 793	11.3
Oklahoma	9	317	0- 943	10.5
Texas	29	1060	0- 3150	7.4
Montana	1	48	0- 142	6.1
Wyoming	1	37	0- 109	7.8
Colorado	8	280	0- 834	9.7
New Mexico	3	109	0- 325	8.4
Idaho	1	46	0- 138	4.9
Utah	3	104	0- 308	7.1
Arizona	5	193	0- 574	7.1
Washington	4	146	0- 434	3.5
Oregon	2	87	0- 257	3.3
Nevada	1	38	0- 114	4.8
California	22	816	0- 2425	3.4

APPENDIX 2

Health Impacts - 1978 Baseline
Canada

State	Sulfate Exposure (million P-ug/m^3)	Excess Mortality (Deaths) Median	90% Range	Mortality Rate (Deaths /100000 P)
Manitoba	2.	63.	0- 189.	6.1
Ontario	57.	2094.	0- 6227.	24.7
Sudbury	1.	27.	0- 82.	27.4
Quebec	34.	1248.	0- 3710.	19.7
New Brunswick	4.	160.	0- 448.	21.2
Nova Scotia and PEI	5.	196.	0- 583.	17.6
Newfoundland & Labrador	1.	51.	0- 151.	8.8
Saskatchewan & Alberta	5.	168.	0- 500.	5.6
British Columbia	2.	79.	0- 236.	3.1
Totals	1372.	50774.	0- 150950.	

APPENDIX 3

Percent Contribution to Sulfate Concentrations in Selected States and Provinces - 1978 Baseline, United States

State	Ohio	Missouri	New York	Wisconsin	North Carolina	South Dakota
Ohio	16.4	2.3	16.8	4.9	11.6	.7
Illinois	7.9	5.0	6.1	10.3	4.7	2.3
Pennsylvania	4.0	.4	11.7	1.0	4.6	.1
Indiana	12.4	5.3	9.0	8.9	7.9	1.6
Kentucky	7.9	3.1	5.1	3.3	8.0	.8
Michigan	4.6	1.6	5.2	3.8	3.1	.8
Tennessee	6.3	3.9	3.9	3.3	7.7	1.0
Missouri	6.0	11.9	3.8	10.7	3.6	4.7
West Virginia	3.9	.6	5.5	1.0	5.3	.2
New York	.9	.1	4.1	.3	1.6	.0
Alabama	3.0	4.6	1.4	2.3	4.4	1.5
Wisconsin	3.0	2.3	2.6	6.2	1.8	1.5
Iowa	1.5	2.1	1.1	4.1	.9	1.5
Minnesota	1.2	2.1	.9	4.8	.7	3.3
Virginia	.8	.1	1.3	.2	2.0	.0
North Carolina	1.4	.3	1.7	.4	4.5	.1
Florida	1.7	1.6	1.2	.9	5.5	.5
Georgia	2.3	1.4	1.5	.9	5.6	.3
South Carolina	1.0	.5	.8	.4	3.0	.1
Maryland	.5	.1	1.3	.1	1.0	.0
Delaware	.1	.0	.3	.0	.2	.0
New Jersey	.3	.0	1.3	.1	.7	.0
District of Columbia	.1	.0	.2	.0	.2	.0
Arkansas	.4	1.4	.2	.6	.4	.5
Louisiana	.6	2.3	.3	.9	.7	1.1
Mississippi	.7	1.7	.3	.7	.9	.6
Massachusetts	.1	.0	.6	.0	.2	.0
Connecticut	.1	.0	.4	.0	.2	.0
Rhode Island	.0	.0	.1	.0	.0	.0
Maine	.0	.0	.1	.0	.0	.0
Vermont	.0	.0	.0	.0	.0	.0
New Hampshire	.0	.0	.2	.0	.1	.0
North Dakota	.3	.7	.2	1.1	.2	2.3
South Dakota	.0	.2	.0	.2	.0	.7
Nebraska	.3	1.1	.1	.8	.2	1.4
Kansas	.9	4.3	.5	2.7	.6	3.8
Oklahoma	.4	2.1	.2	1.1	.2	1.8
Texas	3.2	16.9	1.5	7.0	2.6	12.4
Montana	.1	.3	.1	.5	.1	1.9
Wyoming	.2	.9	.1	.8	.1	4.4
Colorado	.3	1.9	.1	1.0	.2	4.4
New Mexico	.6	4.1	.3	1.7	.6	6.0
Idaho	.0	.1	.0	.1	.0	.3
Utah	.0	.2	.0	.2	.0	1.4
Arizona	1.5	8.7	.8	5.0	1.1	21.2
Washington	.0	.1	.0	.2	.0	1.3
Oregon	.0	.0	.0	.0	.0	.2
Nevada	.0	.1	.0	.1	.0	.4
California	.1	.5	.0	.4	.0	2.7

APPENDIX 3

Percent Contribution to Sulfate Concentrations in Selected States and Provinces - 1978 Baseline, United States (Continued)

State	Wyoming	Washington	California	Ontario	Quebec	New Brunswick
Ohio	.1	.0	.0	13.0	14.2	14.5
Illinois	.4	.0	.0	8.1	6.3	4.8
Pennsylvania	.0	.0	.0	6.2	9.3	11.7
Indiana	.2	.0	.0	9.8	8.6	7.4
Kentucky	.1	.0	.0	4.4	3.9	3.8
Michigan	.1	.0	.0	6.0	5.1	4.0
Tennessee	.2	.0	.0	3.5	3.0	2.8
Missouri	1.0	.1	.0	5.9	4.3	3.0
West Virginia	.0	.0	.0	3.4	4.3	5.1
New York	.0	.0	.0	2.1	4.7	7.5
Alabama	.6	.1	.1	1.5	.9	.7
Wisconsin	.3	.0	.0	4.1	2.9	2.0
Iowa	.3	.0	.0	1.9	1.3	.9
Minnesota	.7	.1	.1	2.3	1.2	.7
Virginia	.0	.0	.0	.6	.8	1.2
North Carolina	.0	.0	.0	.9	1.1	1.5
Florida	.2	.0	.0	.9	.7	.8
Georgia	.1	.0	.0	1.1	.9	1.0
South Carolina	.0	.0	.0	.6	.5	.6
Maryland	.0	.0	.0	.6	.9	1.3
Delaware	.0	.0	.0	.1	.2	.4
New Jersey	.0	.0	.0	.6	1.1	1.9
District of Columbia	.0	.0	.0	.1	.1	.2
Arkansas	.2	.0	.0	.3	.2	.1
Louisiana	.7	.2	.2	.4	.2	.1
Mississippi	.2	.0	.0	.4	.3	.2
Massachusetts	.0	.0	.0	.3	1.1	2.5
Connecticut	.0	.0	.0	.2	.5	.9
Rhode Island	.0	.0	.0	.0	.1	.2
Maine	.0	.0	.0	.1	.4	1.1
Vermont	.0	.0	.0	.0	.1	.1
New Hampshire	.0	.0	.0	.1	.5	.8
North Dakota	.7	.1	.1	.6	.3	.1
South Dakota	.2	.0	.0	.1	.0	.0
Nebraska	.5	.1	.0	.3	.1	.1
Kansas	1.3	.2	.1	1.1	.6	.4
Oklahoma	.8	.2	.1	.5	.2	.1
Texas	7.6	1.9	1.6	2.9	1.5	.9
Montana	1.7	.6	.1	.3	.2	.1
Wyoming	6.8	1.3	.9	.3	.1	.1
Colorado	5.5	.7	1.2	.3	.1	.1
New Mexico	7.2	1.2	2.6	.6	.3	.2
Idaho	.8	1.1	.5	.0	.0	.0
Utah	3.8	2.7	2.8	.1	.0	.0
Arizona	36.1	10.3	24.4	2.0	.9	.6
Washington	2.6	34.5	2.4	.1	.0	.0
Oregon	.4	3.4	.7	.0	.0	.0
Nevada	1.2	2.1	2.9	.0	.0	.0
California	8.3	18.4	56.3	.2	.1	.0

APPENDIX 3

Percent Contribution to Sulfate Concentrations in Selected States and Provinces - 1978 Baseline, Canada

State	Ohio	Missouri	New York	Wisconsin	North Carolina	South Dakota
Manitoba	.3	.5	.2	1.2	.1	2.3
Ontario	.4	.3	.6	.8	.2	.4
Sudbury	2.5	1.2	5.0	3.2	1.8	1.0
Quebec	.4	.2	1.1	.4	.5	.2
New Brunswick	.0	.0	.1	.0	.0	.0
Nova Scotia and PEI	.0	.0	.1	.0	.0	.0
Newfoundland & Labrador	.0	.0	.0	.0	.0	.0
Saskatchewan & Alberta	.2	.9	.1	1.3	.1	6.1
British Columbia	.0	.0	.0	.1	.0	.4

State	Wyoming	Washington	California	Ontario	Quebec	New Brunswick
Manitoba	1.1	.8	.2	1.1	.6	.3
Ontario	.1	.0	.0	1.2	1.2	.7
Sudbury	.2	.1	.0	6.9	9.1	5.7
Quebec	.1	.0	.0	1.1	3.3	3.0
New Brunswick	.0	.0	.0	.1	.8	2.1
Nova Scotia and PEI	.0	.0	.0	.0	.5	1.3
Newfoundland & Labrador	.0	.0	.0	.0	.1	.2
Saskatchewan & Alberta	7.0	9.2	2.0	.7	.3	.1
British Columbia	.7	10.3	.6	.0	.0	.0

APPENDIX 4

Population, Emissions, and Sulfate Concentration - 1978 Emissions
2000 Population

State	Population (millions)	SOx Emissions (1000 tons)	Sulfate Concentration (ug/m^3)
Ohio	11.11	3100.	9.6
Illinois	12.08	1489.	6.2
Pennsylvania	12.30	2067.	10.5
Indiana	5.85	2065.	7.8
Kentucky	4.40	1277.	7.0
Michigan	10.13	988.	7.6
Tennessee	5.41	1116.	5.8
Missouri	5.41	1296.	4.3
West Virginia	2.18	1140.	9.2
New York	17.62	1109.	10.0
Alabama	4.54	694.	3.8
Wisconsin	5.50	672.	5.3
Iowa	3.15	357.	4.2
Minnesota	4.61	400.	3.3
Virginia	6.78	387.	8.9
North Carolina	7.31	573.	7.0
Florida	14.53	927.	2.6
Georgia	7.00	641.	4.5
South Carolina	3.99	287.	5.5
Maryland	5.30	318.	9.6
Delaware	.68	87.	9.5
New Jersey	8.30	367.	9.7
District of Columbia	.58	40.	9.6
Arkansas	2.75	111.	3.4
Louisiana	4.76	258.	2.2
Mississippi	2.86	199.	3.1
Massachusetts	6.60	348.	9.2
Connecticut	3.36	142.	9.4
Rhode Island	1.10	30.	9.0
Maine	1.41	105.	6.8
Vermont	.61	15.	8.7
New Hampshire	1.16	84.	8.3
North Dakota	.73	138.	2.2
South Dakota	.75	26.	2.5
Nebraska	1.82	94.	2.8
Kansas	2.59	327.	3.1
Oklahoma	3.62	158.	2.8
Texas	18.20	1669.	2.0
Montana	.97	148.	1.6
Wyoming	.59	218.	2.1
Colorado	3.96	291.	2.6
New Mexico	1.69	696.	2.3
Idaho	1.25	37.	1.3
Utah	1.92	148.	1.9
Arizona	3.97	2335.	1.9
Washington	4.64	518.	1.0
Oregon	3.27	52.	.9
Nevada	1.05	77.	1.3
California	28.59	794.	.9

APPENDIX 4

Population, Emissions, and Sulfate Concentration - 1978 Emissions
2000 Population, Canada

State	Population (millions)	SOx Emissions (1000 tons)	Sulfate Concentration (ug/m^3)
Manitoba	1.09	442.	1.6
Ontario	9.92	240.	6.7
Sudbury	.12	1114.	7.4
Quebec	6.62	368.	5.3
New Brunswick	.79	221.	5.7
Nova Scotia and PEI	1.06	254.	4.8
Newfoundland & Labrador	.62	61.	2.4
Saskatchewan & Alberta	4.26	642.	1.5
British Columbia	3.20	229.	.8
Totals	290.65	33984.	

APPENDIX 5

Health Impacts - 1978 Emissions, 2000 Population

State	Sulfate Exposure (million P-ug/m^3)	Excess Mortality (Deaths)		Mortality Rate (Deaths /100000 P)
		Median	90% Range	
Ohio	106	3930	0- 11685	35.4
Illinois	74	2753	0- 8184	22.8
Pennsylvania	129	4788	0- 14234	38.9
Indiana	46	1688	0- 5017	28.8
Kentucky	31	1136	0- 3377	25.8
Michigan	77	2860	0- 8504	28.2
Tennessee	31	1159	0- 3447	21.4
Missouri	23	863	0- 2566	15.9
West Virginia	20	743	0- 2209	34.2
New York	176	6522	0- 19388	37.0
Alabama	17	642	0- 1910	14.1
Wisconsin	29	1082	0- 3217	19.7
Iowa	13	487	0- 1447	15.4
Minnesota	15	557	0- 1655	12.1
Virginia	60	2230	0- 6630	32.9
North Carolina	51	1881	0- 5593	25.7
Florida	37	1385	0- 4117	9.5
Georgia	32	1177	0- 3498	16.8
South Carolina	22	810	0- 2407	20.3
Maryland	51	1882	0- 5595	35.5
Delaware	6	239	0- 712	35.0
New Jersey	81	2991	0- 8893	36.1
District of Columbia	6	206	0- 614	35.6
Arkansas	9	346	0- 1028	12.6
Louisiana	11	395	0- 1174	8.3
Mississippi	9	323	0- 961	11.3
Massachusetts	60	2238	0- 6653	33.9
Connecticut	32	1168	0- 3473	34.8
Rhode Island	10	367	0- 1092	33.3
Maine	10	355	0- 1064	25.1
Vermont	5	196	0- 582	32.4
New Hampshire	10	360	0- 1069	30.9
North Dakota	2	61	0- 180	8.3
South Dakota	2	70	0- 209	9.4
Nebraska	5	189	0- 561	10.4
Kansas	8	292	0- 868	11.3
Oklahoma	10	380	0- 1129	10.5
Texas	37	1355	0- 4029	7.4
Montana	2	59	0- 176	6.1
Wyoming	1	46	0- 136	7.8
Colorado	10	384	0- 1142	9.7
New Mexico	4	142	0- 422	8.4
Idaho	2	61	0- 182	4.9
Utah	4	136	0- 404	7.1
Arizona	8	282	0- 839	7.1
Washington	4	184	0- 487	3.5
Oregon	3	107	0- 319	3.3
Nevada	1	50	0- 148	4.8
California	27	985	0- 2930	3.4

APPENDIX 5

Health Impacts - 1978 Emissions, 2000 Population, Canada

State	Sulfate Exposure (million P-ug/m^3)	Excess Mortality (Deaths)		Mortality Rate (Deaths /100000 P)
		Median	90% Range	
Manitoba	2.	67.	0- 198.	5.1
Ontario	66.	2452.	0- 7290.	24.7
Sudbury	1.	32.	0- 95.	27.4
Quebec	35.	1306.	0- 3883.	19.7
New Brunswick	5.	167.	0- 497.	21.2
Nova Scotia and PEI	5.	186.	0- 553.	17.6
Newfoundland & Labrador	1.	54.	0- 161.	8.8
Saskatchewan & Alberta	6.	239.	0- 711.	5.6
British Columbia	3.	98.	0- 293.	3.1
Totals	1544.	57124.	0- 169828.	

APPENDIX 6

Population, Emissions, and Sulfate Concentration - 70% 1978 Emissions, 2000 Population, United States

State	Population (millions)	SOx Emissions (1000 tons)	Sulfate Concentration (ug/m^3)
Ohio	11.11	2170	6.7
Illinois	12.08	1043	4.3
Pennsylvania	12.30	1447	7.4
Indiana	5.85	1445	5.5
Kentucky	4.40	894	4.9
Michigan	10.13	691	5.3
Tennessee	5.41	781	4.1
Missouri	5.41	907	3.0
West Virginia	2.18	798	6.5
New York	17.62	776	7.0
Alabama	4.54	486	2.7
Wisconsin	5.50	470	3.7
Iowa	3.15	250	2.9
Minnesota	4.61	280	2.3
Virginia	6.78	271	6.2
North Carolina	7.31	401	4.9
Florida	14.53	649	1.8
Georgia	7.00	448	3.2
South Carolina	3.99	20[illegible]	3.8
Maryland	5.30	223	6.7
Delaware	.68	61	6.6
New Jersey	8.30	257	6.8
District of Columbia	.58	28	6.7
Arkansas	2.75	78	2.4
Louisiana	4.76	181	1.6
Mississippi	2.86	139	2.1
Massachusetts	6.60	244	6.4
Connecticut	3.36	99	6.6
Rhode Island	1.10	21	6.3
Maine	1.41	74	4.8
Vermont	.61	10	6.1
New Hampshire	1.16	59	5.8
North Dakota	.73	97	1.6
South Dakota	.75	18	1.8
Nebraska	1.82	66	2.0
Kansas	2.59	229	2.1
Oklahoma	3.62	111	2.0
Texas	18.20	1168	1.4
Montana	.97	104	1.1
Wyoming	.59	153	1.5
Colorado	3.96	204	1.8
New Mexico	1.69	487	1.6
Idaho	1.25	26	.9
Utah	1.92	104	1.3
Arizona	3.97	1635	1.3
Washington	4.64	363	.7
Oregon	3.27	36	.6
Nevada	1.05	54	.9
California	28.59	556	.7

APPENDIX 6

Population, Emissions, and Sulfate Concentration - 70% 1978 Emissions, 2000 Population, Canada

State	Population (millions)	SOx Emissions (1000 tons)	Sulfate Concentration (ug/m^3)
Manitoba	1.09	310.	1.2
Ontario	9.92	168.	4.7
Sudbury	.12	780.	5.2
Quebec	6.62	258.	3.7
New Brunswick	.79	154.	4.0
Nova Scotia and PEI	1.06	178.	3.3
Newfoundland & Labrador	.62	42.	1.7
Saskatchewan & Alberta	4.26	449.	1.1
British Columbia	3.20	161.	.6
Totals	290.65	23789.	

APPENDIX 7

Health Impacts - 70% 1978 Emissions, 2000 Population, United States

State	Sulfate Exposure (million P-ug/m^3)	Excess Mortality (Deaths) Median	90% Range	Mortality Rate (Deaths /100000 P)
Ohio	74	2751	0- 8179	24.8
Illinois	52	1927	0- 5729	15.9
Pennsylvania	91	3352	0- 9964	27.2
Indiana	32	1181	0- 3512	20.2
Kentucky	21	795	0- 2364	18.1
Michigan	54	2002	0- 5953	19.8
Tennessee	22	812	0- 2413	15.0
Missouri	16	604	0- 1796	11.2
West Virginia	14	520	0- 1547	23.9
New York	123	4565	0- 13572	25.9
Alabama	12	450	0- 1337	9.9
Wisconsin	20	757	0- 2252	13.8
Iowa	9	341	0- 1013	10.8
Minnesota	11	390	0- 1159	8.4
Virginia	42	1561	0- 4641	23.0
North Carolina	36	1317	0- 3915	18.0
Florida	26	969	0- 2882	6.7
Georgia	22	824	0- 2449	11.8
South Carolina	15	567	0- 1685	14.2
Maryland	36	1317	0- 3916	24.9
Delaware	5	168	0- 498	24.5
New Jersey	57	2094	0- 6225	25.2
District of Columbia	4	145	0- 430	24.9
Arkansas	7	242	0- 719	8.8
Louisiana	7	276	0- 822	5.8
Mississippi	6	226	0- 673	7.9
Massachusetts	42	1566	0- 4657	23.7
Connecticut	22	818	0- 2431	24.4
Rhode Island	7	257	0- 764	23.3
Maine	7	248	0- 738	17.6
Vermont	4	137	0- 408	22.6
New Hampshire	7	252	0- 748	21.6
North Dakota	1	42	0- 126	5.8
South Dakota	1	49	0- 146	6.6
Nebraska	4	132	0- 393	7.2
Kansas	6	204.	0- 607	7.9
Oklahoma	7	266	0- 790	7.3
Texas	26	949	0- 2821	5.2
Montana	1	41	0- 123	4.3
Wyoming	1	32	0- 95	5.4
Colorado	7	269	0- 799	6.8
New Mexico	3	99	0- 295	5.9
Idaho	1	43	0- 127	3.4
Utah	3	96	0- 283	5.0
Arizona	5	198	0- 587	5.0
Washington	3	115	0- 341	2.5
Oregon	2	75	0- 223	2.3
Nevada	1	35	0- 104	3.3
California	19	690	0- 2051	2.4

APPENDIX 7

Health Impacts - 70% 1978 Emissions, 2000 Population, Canada

State	Sulfate Exposure (million P-ug/m^3)	Excess Mortality (Deaths) Median	Excess Mortality (Deaths) 90% Range	Mortality Rate (Deaths /100000 P)
Manitoba	1.	47.	0- 139.	4.3
Ontario	46.	1716.	0- 5103.	17.3
Sudbury	1.	22.	0- 67.	19.2
Quebec	25.	914.	0- 2718.	13.8
New Brunswick	3.	117.	0- 348.	14.9
Nova Scotia and PEI	4.	130.	0- 387.	12.3
Newfoundland & Labrador	1.	38.	0- 113.	6.2
Saskatchewan & Alberta	5.	167.	0- 498.	3.9
British Columbia	2.	69.	0- 205.	2.2
Totals	1081.	39987.	0- 118879.	

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SECTION II HEALTH EFFECTS OF NITROGEN OXIDES (NO_x)

1. INTRODUCTION

Evidence from a variety of sources implicates NO_x as a harmful air contaminant under a variety of exposure conditions. The extent and significance of harm to humans from environmental levels of NO_x are, however, difficult to quantify. Laboratory studies of NO_x exposure provide evidence of changes in lung structure and function that imply the potential for serious long-term health impacts. Additionally, while not clearly established, the possibility that repeated exposures are associated with chronic lung disease and increased "aging" of the lung in adolescence and adulthood cannot be discounted. As yet the results of epidemiological studies do not permit quantifying projections of health damage to human populations from outdoor exposures; extensive further assessment is necessary.

Results from animal studies suggest that both chronic low-level and acute short-term exposures can have adverse effects on health. However, there is insufficient evidence to determine the relative importance for health of peaking exposures versus long-term low-level concentrations of NO_x. NO_x exposure is of special concern for certain particularly sensitive population groups such as infants and asthmatics; health impacts can be most clearly observed in these groups.

NO_x pollution is principally an urban air quality problem, by virtue of the distribution of both sources and receptors (the latter including high densities of sensitive individuals). Nevertheless, rural populations may also be at risk from indoor concentrations, which have been observed to be high in a variety of residential environments. Effects of indoor air pollution have been observed in epidemiological studies of children. Finally, secondary transformation products of NO_x can be transported over large distances contributing to the problem of acid deposition, with potential health and environmental effects that have yet to be adequately quantified. Accordingly, NO_x must be considered as an air contaminant with the potential for both long- and short-range impact, both acute and chronic.

Over 20 million metric tons of NO_x are released into the atmosphere by the U.S. every year (USEPA, 1979) compared to an estimated 48 (USEPA, 1979) to 53 (WHO, 1977) million metric tons worldwide. While this amount is small compared to natural sources (Robinson and Robbins estimate a natural/anthropogenic emissions ratio of 7/1 [USEPA 1977: 5-3]), anthropogenically produced NO_x is responsible for local concentrations 10-100 times greater than normal background levels.

The effects of nitrogen oxides in combination with other pollutants have not been thoroughly investigated here, but such combinations are likely to be detrimental to public health.

2. NO_x CHEMICAL AND PHYSICAL PROPERTIES

2.1 Identification of Substances

The term oxides of nitrogen (NO_x) refers to numerous compounds, including nitric oxide (NO) and nitrogen dioxide (NO_2). From a health risk perspective these compounds are the most important. Nitrogen oxide derivatives include gaseous nitric acid (HNO_3), nitrates (NO_3), and nitrate aerosols, which are formed in the atmosphere from chemical reactions of NO or other NO_x compounds with non-nitrogenous substances. It has been theorized but not convincingly demonstrated (USEPA, 1979, p. 1-5), that nitrosamine is produced in the atmosphere through a series of chemical reactions; small amounts have been detected in the air (Smith, 1980, p. 17).

2.2 Identification of Sources

Concentrations of nitrogen oxides in ambient air result primarily from the combustion of fossil fuels such as coal, oil, and gasoline. Since 1940, nitrogen oxides emissions from both major source categories -- mobile and stationary -- have increased substantially.

The importance of each source category to population exposures and the relative importance of emissions from local vs. distant sources contributing to total NO_x exposure are unknown. In general, emissions from ground-level sources (automobiles) are not thought to be transported long distances as readily as NO_x emissions from tall stacks (utility and industrial sources).[1] Moreover, indoor sources, such as gas cooking stoves, some home heating systems, and cigarette smoking, contribute to population exposure.

2.3 Photochemical Reactions and Chemical Transformations

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[1] While similarities exist between NO_x compounds and other air pollutants known to be transported over great distances, some differences can be noted. The spatial distribution of both ambient concentrations and acid deposition from NO_x differs from that of SO_x because of the following: 1) Sulfur dioxide is capable of forming twice as much strong acid as is NO or NO_2. Thus, nitrogenous species are likely to contribute more to acid deposition near the emission source, while sulfur species affect areas much further away. 2) These are differences in oxidation rate; NO emissions are converted to NO_2 very quickly, i.e., within hours. NO_2 persists in the atmosphere on the order of a day, and is subsequently converted to nitric acid at an average rate of approximately 4%/hr. SO_2 converts to sulfate or sulfuric acid at an average rate of 1%/hr to 4%/hr. (Atmospheric transformation of sulfur and nitrogen oxides has been addressed in Atmospheric Environment, 1981; see also Logan et al., 1981.) 3) Nitric acid tends to remain in the gaseous state, while virtually all sulfuric acid is present as aerosol, which has a much longer residence time.

In the lower atmosphere, typically around urban centers, photochemical reactions involving principally NO, NO_2, and gaseous organic molecules produce a number of reactive species. These species can, in turn, initiate a number of subsequent reactions and by-products. Such reactions are complex and are affected by solar radiation, temperature, and presence or absence of a variety of hydrocarbons. Many secondary pollutants formed from NO_2 are more toxic on a unit concentration basis than NO_2.

Up to half the oxides of nitrogen may be initially converted to peroxyacetyl nitrate (PAN), but ultimately, formation of nitric acid is likely. Diurnal photochemical modeling also predicts that up to half (and probably more) of the NO_x may be more directly converted to nitric acid and nitrates. Near certain industrial areas, nitrosamines, amides, and other photochemical by-products of nitrosamine formation may be found in sufficient concentrations (5-150 ppb) in polluted urban air to warrant concern for health (USEPA, 1979, p. 6-66).

The amount of NO_x in ambient air can also affect the production of other compounds of concern to health -- produced either directly from nitrogenous reaction (i.e., ozone) or indirectly from non-nitrogenous reactions (i.e., SO_2 oxidation).

3. HEALTH EFFECTS OF NO_x

3.1 Mechanisms of NO_x Toxicity

This section deals predominantly with health effects of NO_2, which has received the most attention in biomedical research. The effects of NO_2 may be attributed to several mechanisms:

1. Formation of acid solutions in the respiratory tract, which probably contribute significantly to the acute irritant effects attributed to NO_2.

2. Oxidation of major constituents of biological membranes, resulting in: a) impaired biochemical and functional properties of cell membranes; b) altered protein conformation. The functional and structural consequences of these reactions are characteristically delayed in appearance; they may be only partially reversible; and c) reaction with hemoglobin to produce methemoglobin; the latter is ineffective as a carrier of oxygen.

3. NO_2 in aqueous solution may also form nitrosamines, which have high carcinogenic potential.[2]

3.2 Animal and In Vitro Research

Studies of animals and isolated tissues have provided information on: 1) the cells, subcellular components, and biochemical systems that are vulnerable to low concentrations of NO_x (principally NO_2); 2) the functional and clinical consequences of such vulnerability; and 3) the importance of different patterns of exposure (e.g., acute, subacute, chronic, intermittent high concentrations imposed on backgrounds of low concentrations) in determining the severity and persistence of any adverse effects that may occur. Results from animal studies confirm that the respiratory system is the principal although not exclusive target of NO_x inhalation. The entire length of the tracheo-bronchial system is subject to irritation and toxicity (implying biochemical or structural injury) from NO_2, depending on the dose. Systemic effects may also be noted.[3]

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[2] The formation of nitrosamines is pH-dependent and is likely to proceed at a faster rate in the strongly acid gastric juice than in the slightly alkaline liquid lining of the lung.

The capacity of the respiratory system to ward off infection may be decreased by exposure to NO_x since the pollutant adversely affects each of the four mechanisms upon which resistance depends.[4] NO_x exposure inhibits mucociliary clearance (the mechanical function of lung clearance by cilia), phagocytosis (the ingestion of foreign bacteria by body cells), and immunological responses (operating both intra- and extra-cellularly).

Laboratory experiments on animals and tissues suggest more serious consequences of long-term human exposure to NO_2 than have appeared in short-term clinical studies. Prolonged exposure to NO_2 has been observed to cause damage to lung tissue in laboratory animals.[5] The principal consequences of such damage appear to be development of emphysema-like conditions, and reductions in resistance to respiratory infection. For a fixed dosage, greater concentrations have been shown to cause greater increases in mortality rates for animals exposed to respiratory infection than greater duration of exposure. This suggests that fluctuating levels of NO_2, such as are found in community air, may prove more toxic than sustained levels of the gas.

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[3] Indirect mutagens require chemical transformation to become effective. NO_2 is not mutagenic. However, it may react with other pollutants to form mutagens or to convert indirect mutagens to direct mutagens.

[4] Giordano and Morrow, 1972. Gardner et al., 1977. Holt et al., 1979. Goldstein, E., Eagle, M.C., and Hoeprich, 1973.

[5] In one study (Gillespie and Berry, 1980), beagles were exposed to NO_2 and NO for 68 months. Functional deterioration was more pronounced in the two years following exposure than in equivalent periods during the exposure.

3.3 Clinical Research

Controlled exposures of human volunteers to NO_x (NO_2, nitrate aerosols, or nitric acid) have been confined to periods lasting up to several hours, administered once or on several successive days. Such studies cast little light on the role of NO_x in the pathogenesis of chronic disease. Functional and biochemical recovery tends to be rapid, and it is not known whether repeated ambient exposures of this type act cumulatively or predispose the lung to permanent damage. Instead, these studies are useful in describing short-term, reversible effects that may aggravate preexisting functional disorders (primarily cardiopulmonary) or reduce the resistance of the respiratory system to infection. Their focus has been primarily on the following:

- The effects of low concentrations of NO_2, nitrate aerosols, or nitric acid on lung function. The subjects have been healthy adults and patients with asthma or bronchitis. Exposures have been carried out with the subjects at rest or exercising. Healthy subjects or patients with bronchitis do not show functional impairment at or below 1.5 ppm NO_2; some asthmatics have experienced symptoms of irritation during exposure to 0.5 ppm (Kerr et al., 1979). $NaNO_3$ and NH_4NO_3 aerosols in concentrations up to 3 mg/m^3 have produced no functional changes in normal or asthmatic adults. Similarly, in one study of 5 healthy adults, 1.5 ppm HNO_3 administered for 10 minutes has had no functional effect (Sacker and Ford, 1981)[6]

- The effects of low concentrations of NO_2 (0.1 ppm or higher) on airway reactivity, particularly among asthmatics. Hypersensitivity of the airways to non-specific environmental stress such as cold air or air pollutants is characteristic of asthma. Concentrations of NO_2 as low as 0.1 ppm may aggravate this hyper-reactive state in some asthmatic individuals. (Orehek et al., 1976).

3.4 Epidemiological Research

Effects on children have been a principal source of information about the hazard posed by NO_2 to health.[7] The nature of the evidence has been two-fold:

1. Changes in incidence of acute respiratory illness, which is presumed to be infectious in origin (Melia et al., 1977, Speizer et al., 1980). This evidence agrees with experimental results in animals showing that NO_2 reduces resistance to infection.

2. Changes in pulmonary function (Speizer et al., 1980). It remains uncertain whether these effects constitute significant risk factors

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[6] The recommended threshold limit value (TLV) for occupational settings, based on an 8-hour time-weighted average sample, is 2 ppm.

[7] There may be several reasons for increased vulnerability in childhood and infancy, including anatomic and biochemical immaturity, greater airway reactivity, and greater ventilation, corrected for body weight, compared to adults. Early life is characterized by obligatory oral breathing, which by-passes the more effective nasal scrubbing surface. Finally, children are less subject to confounding factors such as smoking and occupational exposure; tobacco smoke contains significant amounts of NO_2 and NO, as do some occupational settings.

that affect the growth and development of the lung or contribute to chronic lung disease in adulthood.

Most information implicating NO_2 as a risk to health appears to have been obtained in studies of indoor air pollution, involving homes with unvented gas cooking stoves, rather than studies of community air pollution. Perhaps the most sensitive individuals in the population, in terms of increases in occurrence of acute respiratory illness, are infants under two years of age; adults appear to be less vulnerable than children (Keller et al., 1979).

Early studies of Chattanooga school children (Shy et al., 1970; Pearlman et al., 1971), which suggested an association between ambient concentrations of NO_2 and both altered pulmonary function and acute respiratory illness, are now considered inadequate for establishing dose-response relations owing to their reliance on faulty air monitoring. In a study of Tokyo schoolchildren (Kagawa and Toyama, 1975), negative correlations were found between several pollutants -- NO_2, NO, SO_2, hydrocarbons, and particulate matter -- and lung performance, suggesting narrowing of small airways. The approximate range of hourly concentrations of NO_2 at the time the tests were performed was 40-360 ug/m^3 (0.02-0.19 ppm). To what extent NO_2 and NO among these several pollutants may have been responsible for the observed changes in pulmonary function is conjectural.

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SECTION III INDIRECT HEALTH EFFECTS OF ACID PRECIPITATION

1. INTRODUCTION

Indirect effects of acid precipitation result from the corrosiveness of low pH precipitation and runoff, which may dissolve toxic metals (aluminum, cadmium, copper, lead, mercury), or release such other materials as asbestos, from soils and rocks in the watershed and lakes, and from drinking-water distribution systems. These toxic substances may be ingested with drinking water; additionally, these compounds can be biologically accumulated in the food chain and then ingested by humans consuming fish from affected areas. Toxic substances in drinking water may cause chronic, possibly widespread, exposures.

Toxic heavy metals (arsenic, cadmium, lead, mercury, nickel, zinc) may be deposited directly from the atmosphere independently from acid precipitation and may also find their way into drinking water supplies. Because evidence for elevated drinking water concentrations due primarily to direct atmospheric deposition of metals has not been found, health effects from this pathway will not be discussed further.

Studies which attempt to establish a direct relationship between acid deposition and degraded drinking water quality are rare. Studies which may be used for establishing that relationship indirectly are more numerous but do not demonstrate causality. Establishing such a relationship is beyond the scope of this effort; reference to such studies is solely to identify, for a health effects discussion, those materials in drinking water whose presence may be influenced by acid deposition.

Several criteria were used to select contaminants for discussion in this report. A contaminant must have the potential to be corroded from drinking water distribution systems (asbestos, cadmium, copper, lead) or the watershed (aluminum, asbestos, copper, lead) by aggressive waters. Because asbestos and lead can have severe health effects for potentially large populations, they are discussed in some depth. Cadmium occurs in drinking water only infrequently, and is a potential product of plumbing corrosion. It is considered because of its severe occupational health effects. The health effects of copper and aluminum contamination are less substantial, and affect fewer people.

Acid precipitation contributes to the toxic material burden of drinking water. Although no substantiated illnesses are known to have been caused by interactions between acid precipitation and heavy metals, recent data demonstrate, for certain areas characterized by acid precipitation, low pH, and low-alkalinity drinking water, that drinking water criteria for toxic heavy metals are being exceeded in the United States (Fuhs, 1979; Fuhs and Olsen, 1979; Fuhs et al., 1980; Morse et al., 1979; and others) and in southern Scandinavia (Hultberg and Wenblad, 1980). Whether the lack of reported illness is caused by an actual absence of health effects, by failure to recognize or report occurrences of heavy metal toxemia, or by as-yet-unreached time or concentration exposure thresholds is not known. However, widespread distribution of areas sensitive to acid precipitation, and the spread of acid precipitation to encompass virtually all sensitive areas east of the Mississippi River (Glass, 1980), requires that current knowledge of the distribution of toxic metals in drinking water, and of their related health effects, be examined and expanded.

Two preliminary investigations directly address effects of acid precipitation on increasing toxic heavy metal concentrations in U.S. drinking waters (Fuhs, 1979; Fuhs and Olsen, 1979; Fuhs et al., 1980); the results from these studies are inconclusive. With the exception of the Saranac Lake study, which yielded one violation of copper level standards in an area with a relatively large population exposure, the number of water samples taken and the potentially affected populations are each quite small. A number of studies have been made in the Adirondack region of New York by the New York State Department of Public Health. Sampling locations were diverse (homes, municipal water supply lakes, private wells, a spring), as were the types of water distribution systems sampled (copper, iron, plastic piping).

Acid leaching of plumbing and of granitic Adirondack bedrock was implicated as a potential source of elevated copper, lead, and aluminum concentrations in the drinking waters sampled from wells and shallow springs in the Old Forge-Eagle Bay areas (Fuhs and Olsen, 1979; Fuhs et al., 1980). Copper, lead, and aluminum were measured at the tap at levels reaching 7.2, 0.44, and 1.0 mg/l respectively. Copper exceeded the domestic water quality criterion by a factor of 7 and lead exceeded the lead criterion by a factor of 100. The contribution of atmospheric deposition of lead, and possibly copper, to local watersheds was also indicated.

Studies relating acid groundwater and elevated metal concentrations in southwestern Sweden to short- and long-range transport of sulfur compounds and subsequent acid precipitation (Hultberg and Wenblad, 1980) support the Adirondack results -- showing markedly elevated groundwater and tap water concentrations of aluminum and copper -- and extend these observations to include elevated tapwater concentrations of cadmium and zinc. One study reported groundwater (sampled at depths of 0.5-17m along the length of an acid stream -- pH approaching 3) with aluminum concentrations exceeding 40 mg/l.

Hultberg and Wenblad also reported water quality results from surveys of predominantly shallow (1-10 m) wells in three counties in southwestern Sweden. The main source of acid precipitation in the study areas was long-range transport of sulfur from other countries. Of the 494 sampled wells, 315 had pH values below 6.0. In one county 49% (137) of the wells had a pH below 5.5 and 13% (36) of the owners reported severe corrosion of copper water pipes. Tap water analyses revealed high aluminum (0.4-1.7 mg/l), copper (1-2 mg/l), zinc, "and to some extent" cadmium concentrations which varied as a function of groundwater pH.

The studies by Fuhs and Olsen were only preliminary but indicate several violations of drinking-water criteria. Although drinking-water acidity was not universally high, it reached a pH of 4.95. The Langelier index indicated potentially corrosive drinking water in several locations where metal concentrations were not yet markedly elevated. However, the samples most corrosive had copper and lead concentrations that exceed domestic drinking water criteria values (copper 1.0 mg/l; lead, 0.05 mg/l) by factors as great as 7 and 3, respectively. Even though no deleterious health effects have been associated with these metal concentrations, the potential for metal toxemia exists. Based on the low population density of the study area and the small area sampled, however, the population at risk is small.

To describe more accurately the extent of the problem and the size of the exposed population, the New England Water Works Association, in conjunction with EPA, is studying effects of corrosion on water quality in water distribution systems of New England and the Adirondack region of New

York. Preliminary analysis of the partially completed study does not indicate corrosion of hazardous concentrations of toxic metals from distribution systems.

2. CORROSIVENESS OF WATER

Several indices have been developed to characterize the potential corrosiveness of water to distribution systems. Although corrosion indices are frequently used to assess theoretical potential for corrosion, several factors limit their accuracy. Actual corrosion depends on the conditions within the system when corrosive water contacts the pipes. Water distribution systems are protected from corrosion by layers of scale, a protective film deposited on the interior of the pipe from salts dissolved in the distributed water. Calcium carbonate ($CaCO_3$) is a major constituent of scale. Formation of a calcium carbonate film depends on independent and interactive actions of pH, alkalinity, hardness, total dissolved solids, and temperature.[1]

Corrosion problems are dynamic and specific to each water distribution system. They depend on interactions between water quality parameters, water treatment procedures, and occurrence, spatial distribution, and age of construction materials in the distribution system.

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[1] Alkalinity of water is a measure of its capacity to neutralize acids, and is related to concentrations of carbonate, bicarbonate, and hydroxide. Total alkalinity is usually expressed in terms of the equivalent amount of calcium carbonate. Hardness is also expressed in terms of the equivalent amount of calcium carbonate and represents principally the concentrations of calcium and magnesium, but also includes other dissolved metal ions (e.g., aluminum, iron, manganese, and zinc). For example, medium-hard water has 61-120 mg/l total hardness as $CaCO_3$; soft water has less, and hard water has more. Scale-forming ability decreases, and hence potential corrosion increases, with decreasing hardness and alkalinity. A high concentration of total dissolved solids increases the conductivity of water and encourages corrosion. Increasing temperature and oxygen concentrations also increases the corrosive tendencies of water.

2.1 Potentially Corrosive Drinking Water: An Example

Essentially all of eastern North America receives acidic precipitation below pH 4.6 (weighted mean, annual average 1979, U.S. Canada 1981). Although detailed analysis of specific water supplies is beyond the scope of this report, researchers have examined an "Aggressiveness Index" for drinking water from 153 Massachusetts reservoirs, indicating that Massachusetts water supplies are highly aggressive.[2] Data from eight (8) Metropolitan District Commission reservoirs (including the Quabbin reservoir, which provides about 60% of the water for the Boston Metropolitan Water District), indicate that a large population is receiving aggressive drinking water, some transported over great distances, capable of corroding asbestos from asbestos-cement (A/C) pipe, or toxic metals from lead, lead-lined, or galvanized distribution pipes and plumbing.

3. ASBESTOS

Asbestos fibers may enter water from natural erosion of asbestos-containing rock, mine processing wastes, waste pile erosion, corrosion of asbestos-cement pipe,* or disintegration of asbestos tile roofs used to collect and direct rain water into drinking water cisterns.

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[2] Data provided by Alan Van Arsdale, Massachusetts Department of Environmental Quality Engineering. Two caveats concerning this calculation are appropriate: 1) each reservoir is represented by only one sampling date and 2) spring sampling dates may have depressed pH's caused by sudden influx of hydrogen ions if sampling occurred during snow melt.

* Asbestos fibers, mixed in concentrations of 10 to 25 percent with cement, provide a matrix which binds and reinforces the finished asbestos-cement product.

Table 1
Selected Asbestos Concentrations
Found in Some Drinking Water Systems[1]

	Range of Asbestos Concentration 10^6 fibers/liter
Asbestos - cement pipe	
Bishopville, SC	0.4-500
Kentucky, Dam Village, KY	45
Lakeland, FL	0.2-17
Pensacola, FL	0.05-32
Paint, PA	4-19
San Francisco, CA[2]	0.03-8.6
Asbestos - cement tile roof[3]	
Cistern, St. Croix, V.I.	15-543
Cistern, St. Thomas, V.I.	20-30
Cistern, Oxford, OH	96

1. Data from Millette et al, 1981a, unless otherwise noted.
2. Data from Kanarek et al, 1980.
3. Data from Millette et al, 1980b.

3.1 Estimates of U.S. Exposure to Corrosive Water and Asbestos

U.S. exposure to asbestos in drinking water from distribution pipes was estimated by Millette (Millette et al, 1981a; 1981b). (See Table 1) Available data indicates that 16.5% of U.S. water utilities distribute highly aggressive water, and 52% distribute moderately aggressive water.[3] Based on available data from 426 selected U.S. water suppliers, some of which were analyzed because of suspected asbestos contamination, Millette has estimated that approximately 11% of the U.S. population is exposed to asbestos concentrations greater than 10 million fibers per liter.

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[3] Between 84,000 and 200,000 miles of A/C drinking water distribution pipe are used in the U.S. Based on an estimate of 300 persons served per mile of pipe, and 200,000 miles of pipe, an estimated 10 million people are served by A/C pipe carrying highly aggressive water and about 30 million are served by moderately aggressive water.

3.2 Health Effects of Asbestos

No data directly associates acid precipitation with asbestos-related health effects. There are several explanations for this lack of association: diagnosis of asbestos-related cancer is difficult (Kannerstein and Churg, 1980); study of acid precipitation health effects and asbestos contamination of drinking water are each recently-discovered concerns and no experimental attempt has yet been made to associate them; a time lag may be involved between exposure of material to aggressive water and corrosion onset or detection; a 20-30 year time lag occurs between occupational exposure and detection of asbestos-associated disease.

Because asbestos is a suspected human carcinogen, and there is no recognized safe concentration for a human carcinogen, the recommended concentration of asbestos in water for maximum protection of human health is zero. Cancer risk levels resulting from various asbestos concentrations, assuming an incremental risk from consuming ambient drinking water only, have been calculated and published in the Federal Register (EPA, 1979). Contributions from other exposure sources are ignored. Risk calculations were based on a daily consumption of 2 liters of drinking water over a 70 year life span. A risk of 10^{-5}, for example, indicates the probability (1 in 100,000) of one additional cancer for each 100,000 exposed people, at the specified asbestos concentration. (See Table 2)

TABLE 2[1]

Cancer Risk Levels From Exposure To Asbestos In Drinking Water

Exposure Assumption	Risk Levels and Corresponding Asbestos Concentrations 10^{-7}	10^{-6}	10^{-5}	10^{-4}
2 liters of drinking water daily for 70 years	3,000 f/l*	30,000 f/l	300,000 f/l	3,000,000 f/l

1. Federal Register 44 (191) October 1, 1979, pg. 56634.

* f/l = fibers/liter

3.3 Epidemiology of Drinking Water Asbestos and Cancer

Kanarek et al. (1980) have recently published the first study concluding that a statistically significant association ($P<0.01$, t-test for multiple regression and correlation coefficients) exists between asbestos in drinking water and cancer in certain body parts. The water comes from aquifers or is stored in reservoirs which have variable contact with naturally occurring asbestos rock. Their findings are consistent with earlier results of Wigle (1977) for Quebec, and Levy et al. (1976) for Duluth.

4. LEAD

Concern about lead and public health effects is derived principally from dissolution of lead (Pb) from lead used in domestic plumbing and water distribution systems -- an application currently prohibited by most U.S. building codes. The solubility of lead is greatest in areas characterized by aggressive waters -- those low in pH, alkalinity, and mineral content (hardness). The current lead ambient water quality criterion for protection of human health is 50 micrograms of lead per liter (ug/l) of drinking water.

Lead enters aquatic ecosystems from wet and dry precipitation, mineral erosion, street runoff, and municipal and industrial discharge (EPA, 1980b). The solubility of lead in water ranges between about 10,000,000 ug/l at pH 5.5 and 1 ug/l at pH 9. Corrosion of lead from lead-containing materials is greatly facilitated by low pH (<6.5) soft waters (EPA, 1980b). Although pH is clearly important in determining corrosion of lead from lead-containing material, e.g., plumbing, a strong cause-effect relationship between precipitation pH and lead corrosion has not been firmly established. Circumstantial evidence for such a relationship is strong.

Mushak (1981) recently reviewed several reports of elevated lead in drinking water. Generally the highest lead concentrations were found in "first-draw" morning samples, and surveys encompassing wide geographic areas show relatively few exceedances of the lead water quality criterion. Durfor and Becker (1964) surveyed 1962 water samples for 100 of the largest U.S. cities and found an average lead concentration of 3.7 ug/l (range 0-62 ug/l). McCabe et al., (1970) reported that in 1969 1.4% of 2595 samples from 969 municipal water supplies exceeded 50ug/l (mean: 13 ug/l; range: 0-640 ug/l).

In contrast, in a study of tap versus central water supply, Craun and McCabe (1975) reported that 76% of Seattle, Washington samples and 65% of Boston, Massachusetts samples exceeded 50 ug/l. In a second study of the Boston area, 50% of first-drawn samples exceeded 50 ug/l, with a mean value of 30 ug/l for 383 households (Karalekas et al., 1975). Worth (1975) reported that 60% of Boston households had some lead plumbing. Morse et al., (1979) reported an average water lead content ratio of 48 ug/l for Bennington, Vermont. About one-half of the 105 surveyed homes had lead plumbing, and of these about one-half had drinking water lead concentrations in excess of 50 ug/l. Bennington receives low-pH precipitation (4.3 - 4.4, weighted mean, annual average), and draws its water from a lead-free mountain brook of pH 5.5 - 6.5 and low alkalinity and hardness. Elevated lead in drinking water is a locale-specific problem related to local water chemistry; and geological, precipitation chemistry, and lead availability factors.

Karalekas (1980) reported water treatment studies conducted in Somerville, Boston, and New Bedford, Massachusetts and Bennington, Vermont for control of lead corrosion. Three treatment processes were used which proved to be effective. The effective treatments involved raising the pH and alkalinity of the water with either sodium hydroxide (NaOH), a combination of NaOH and sodium bicarbonate ($NaHCO_3$) or sodium carbonate (Na_2CO_3). In the systems studied, the maximum reduction in lead concentration was achieved when pH was raised above 8.0.

4.1 Sources of Lead in Drinking Water

A definite positive correlation exists between lead concentration in domestic water supplies and lead concentrations in blood (PbB), the principal index of lead exposure (EPA, 1980b). (See Table 3) A curvilinear relationship exists between water-lead and blood-lead levels; as water-lead levels increase, corresponding increments in PbB decrease.

Table 3
PbB Levels Due To Water Lead

Running Water (ug/l)	PbB Due to Water (ug/dl)			
	Greathouse and Craun, 1976	Moore, et al.[a] 1977a	Hubermont, et al.[a] 1978	All 3 Studies
1	2.54	3.41	0.83	2.26
5	4.35	5.82	2.05	4.07
10	5.47	7.34	2.57	5.13
25	7.43	9.96	3.26	6.88
50	9.36	12.55	3.79	8.57
100	11.79	15.81	4.31	10.64

a. These values were all calculated using morning or "first flush" water values which were taken to be three times the running water levels in the table. The EPA assessment of the contribution of lead in drinking water to PbB was based on studies by Elwood et al., (1976) for first draw tap water in northwest Wales, by Moore, et al., (1977a) for Scotland, by Hubermont et al., (1978) for pregnant and rural dwelling women, and by Greathouse and Craum (1976) for Boston.

From EPA (1980b) Ambient Water Quality Criteria for Lead, EPA 440/5-80-057.

4.1 Distribution of Absorbed Lead

The major body sink of lead is bone. Lead is toxic to the hematological system, the nervous system, and the kidney. Pathological nervous system effects of lead include encephalopathy and more subtle behavioral effects. Such effects are characteristic of adults with occupational exposure, and of children whose exposures were due primarily to pica and ingestion of lead-containing paint chips.

Preliminary human (Beattie, et al., 1975; Moore, et al., 1975b) and animal studies (Carson, et al., 1974; Crofton, et al., 1978) have suggested that in vivo fetal lead exposure is considerably more neurotoxic than exposure to the infant or young child.

Renal function is impaired at PbB levels of approximately 130 ug/dl (Waldron, 1980). Kurzel and Cetrulo (1981) reviewed the effects of lead on the reproduction system and conclude that it is both toxic to embryos and causes teratogenesis (fetal anomalies).

Populations at Risk

EPA has defined approximately 2,500,000 people -- inner-city children under five and pregnant women -- as being at special risk from elevated levels of lead in the bloodstream. Previous investigations have established that inner-city residents, particularly children, have the greatest risk from multi-media lead exposure. Urban rather than rural populations have been assessed for several reasons. Urban populations are served by municipal water supplies where water quality records are more readily available than rural, possibly private supplies.

However, elevated blood lead levels may constitute a greater threat to rural populations using acidified groundwaters than to urban populations using waters that can readily be treated. Preliminary investigations in the New York State Adirondacks region have found isolated cases of human exposure to lead-contaminated drinking water resulting in lead concentrations in blood as high as 59 ug/dl.[5]

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[5] New York State Department of Health, Technical Memorandum: "Lead Poisoning Related to Private Drinking Water Sources," October 2, 1981.

5. CADMIUM

Cadmium is a relatively innocuous material in its natural state. It is a non-essential element for human metabolism (Lauwerys, 1978) and, in concentrated or pure form, is quite toxic. Because approximately 70% of total world production has occurred within the past 20 years, there is growing concern that, without proper precautionary measures, the use of cadmium may involve a potential health hazard both now and in the future.

5.1 Potential Sources of Cadmium in Drinking Water

In nature cadmium does not occur in a pure state; it is always present as an impurity in zinc, bringing it into contact with drinking water through water pipe, solder, and alloys (e.g., brass and bronze).

Little current evidence of cadmium corrosion from galvanized pipe or plumbing alloys exists. Cadmium corrosion may well be time dependent. After a protective film has been formed within the pipe, the cadmium is isolated from the water and corrosion may cease or be greatly reduced. If a film does not form, and the zinc galvanizing, with cadmium contaminant, is subject to continued corrosion, the galvanic coating (and cadmium) may be corroded within a variable period of time on the order of several years, depending on the water chemistry, particularly the carbonate content.

A nationwide sampling program was reported by McCabe et al. (1970). Cadmium levels exceeded the 10 ug/l standard in 0.2 percent of 2595 samples from 969 water supplies. The maximum recorded concentration was 11 ug/l.

5.2 Metabolic Effects

The toxic effects of cadmium in man are of two types: acute manifestations following a high level of Cd exposure over a short period, and symptoms precipitated by long-term or chronic intake. Exposure to cadmium in drinking water would be a long-term effect. The lungs and gastrointestinal tract are the two main routes of entry. Inhalation is the more efficient mode of absorption and is of primary interest from the occupational standpoint, but except for cigarette smoking, ingestion is the major contribution to total body burden for the general public.

Based on the infrequent occurrence of cadmium in drinking water at levels above 10 ug/l, cadmium is considered a potential threat to relatively few people. On the basis of various literature reports, Lauwerys (1978) identified a rough scale of acute oral toxicity of cadmium in man ranging from 3mg (nausea) to >4000mg (lethal). These doses are several orders of magnitude greater than would be received as corrosion products from pipes. No chronic poisoning from ingestion of cadmium and its compounds is reported in the literature.

6. MERCURY

Studies in Scandanavia, Canada, and the U.S. have found correlations between elevated levels of mercury in fish and increasing acidity of lake and stream habitats. However, no cases of mercury poisoning are known to be associated with freshwater fish in the U.S. Lakes in acid-sensitive regions of Minnesota and in the Adirondacks of New York have yielded fish with mercury concentrations above the FDS public health standard of 0.5 ppm, prompting Minnesota officials to issue consumption advisory warnings for many lakes, and the New York State Health Department to advise residents to eat freshwater fish only once a week, and pregnant women and nursing mothers not to eat freshwater fish.[6] Mercury levels which violate the FDA standard have recently been found in acid-sensitive lakes in Wisconsin; studies are currently underway to assess correlations between observed mercury levels and lake pH levels.[7]

Mercury can enter the atmosphere through several pathways, including the combustion of mercury present in coal and other fossil fuels; emissions from chemical processing, sewage plants, and incinerators; and emissions from volcanoes and other natural sources.

A number of recent efforts to neutralize acidic lakes through the application of lime appear to be successful in reducing mercury levels in fish. Favorable results have been reported for various locations in Sweden, and for Nelson Lake in Ontario, Canada.[8]

7. OTHER METALS OF CONCERN

7.1 Aluminum

Little is known about the toxicity of aluminum to humans. The United States has no restrictions on aluminum concentrations in domestic drinking water or foods. Recent studies, however, indicate that aluminum can be highly toxic under specific circumstances (Alfrey et al. 1976, 1980; Wing et al., 1980). Persons with proper renal function who ingest large amounts of aluminum absorb generally small amounts and subsequently excrete most of the element. However, elimination in uremic patients is impaired -- depending on dosage, aluminum concentrations increase markedly in bone, brain, heart, liver, lung, spleen and parathyroid gland. Aluminum accumulation by the brain is associated with the neurological symptoms of speech disorder, and progressive dementia. Water containing >50 ug/l of aluminum is considered unsafe for dialysis treatment (Cann et al., 1980).

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[6] Acid Precipitation in Minnesota, Minnesota Pollution Control Agency, Minnesota Department of Natural Resources, Minnesota Department of Health, January 26, 1982, p. 7-67; Letter from G. Wolfgang Fuhs, Director, N.Y.S. Department of Health, to Congressman Toby Moffett, July 16, 1980, p. 3.

[7] Personal communication, James Wiener, U.S. Fish and Wildlife Service, La Crosse Field Research Station, La Crosse, Wisconsin, May 1982.

[8] "The Role of Clouds in Atmospheric Transport of Mercury and Other Pollutants," G. H. Tomlinson, R. J. P. Brouzes, R. A. N. McLean, and John Kadlecek, in Ecological Impact of Acid Precipitation, Proceedings of an International Conference, Sandefjiord, Norway, March 11-14, 1980, Oslo-As.

The highest aluminum concentrations found in rural drinking water by Fuhs et al., (1980) are 1.0 and 1.4 mg/l; and Hultberg and Wenblad (1980) report well-water concentrations of aluminum reaching 1.7 mg/l in Sweden. Such high aluminum concentrations could represent a substantial proportion of an individual's average daily aluminum intake (4-52 mg/day, Underwood, 1971). Because so little is known about aluminum toxicity, care should be taken in exposing populations to greater than normal dietary aluminum exposure.

7.2 Copper

Copper is an essential element for plants and animals. A limit of 1 mg/l for copper concentration in domestic drinking water was set on the basis of taste rather than toxicity (EPA, 1976, NAS, 1977); there is no legislated limit for copper in foods. For normal human beings, copper deficiency is more widespread than copper toxicity. However, there are several reported cases of toxicity among individuals with Wilson's disease, a formerly fatal inherited disorder of copper metabolism. Patients with Wilson's disease accumulate large amounts of copper in the kidney, brain, and cornea.

Copper levels in several of the drinking water samples analyzed by Fuhs and Olsen (1979) exceeded the EPA guidelines of 1 mg/l. The average daily ingestion of copper in foods is 2-5 mg. Persons consuming water containing 6-7 mg/l of copper could easily exceed the average daily copper intake. However, except for the unlikely possibility of potentiating latent Wilson's disease, the hazard to the general population from these observed copper levels is small.

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MECHANISMS FOR THE CONTROL OF INTERSTATE AND TRANSBOUNDARY AIR POLLUTION

CHAPTER ONE: AVENUES FOR THE CONTROL OF INTERSTATE POLLUTION UNDER THE CURRENT CLEAN AIR ACT.

I. Overview

Much attention has been focused on Congressional proposals for revising the Clean Air Act to control long-range transport air pollutants (LRTAPs). The present Clean Air Act contains certain provisions for addressing interstate air pollution. These provisions rely on the state implementation plan (SIP) process to control interstate pollution through national air quality standards. 1/ Actions taken under these provisions suggest that the Act has been interpreted by EPA and the Courts to provide adequate remedy for interstate air pollution from local sources. However, considerable uncertainty exists over how EPA and the Courts will interpret the provisions with regard to LRTAPs. EPA currently takes the position that analytical techniques addressing LRTAPs are not reliable for regulatory purposes; the courts have yet to rule on this issue.

The Clean Air Act 2/ currently contains two means of remedy for interstate air pollution. The first relies on the general judicial review provisions of the Act which provide for review of EPA actions. These provisions allow plaintiffs to challenge, for instance, the EPA Administrator's approval of SIP revisions, on the basis that the SIPs violate the interstate requirements of section 110(a)(2)(E). The second relies initially on the administrative review provision of section 126 which allows states and political subdivisions to petition EPA on matters dealing specifically with interstate pollution. A number of suits and petitions have been filed under these sections of the Act since 1977 to remedy interstate air pollution. The actions before the courts and EPA can roughly be categorized according to the type of pollution they seek to remedy. Early actions sought remedy for interstate air pollution in situations where pollutants were traceable to individual sources in the near vicinity. In later actions, attention has turned from the "local source problem" and focused on interstate pollution caused by long-range transport. In these actions, some states and interest groups have argued that the total loadings of all the sources within "upwind regions," as well as individual sources, are causing violations of NAAQS and PSD increments, threats to health, and damage to ecosystems as a result of acidic deposition in "downwind states". They also assert that the present EPA interpretation of the section of the Act dealing with interstate pollution (section 110(a)(2)(E)) is inadequate to comply with the requirements of the Clean Air Act and to control interstate pollution problems caused by long-range transport. 3/ Petitioners have requested the courts and EPA in suits in section 126 proceedings to remedy pollution from multiple as well as single sources. They have also sought to require EPA to develop regionally-applicable policies to address long-range transport.

The two means of remedy for interstate pollution -- judicial and administrative -- may differ as to the type of review which can be offered. EPA has asserted, and the 2nd Circuit Court of Appeals has ruled, that EPA is not required to review the whole SIP for its compliance with the Act when EPA reviews proposed SIP revisions which only amend portions of the plan. Accordingly, plaintiffs might not be able to obtain review of the SIP as a whole in the context of SIP revisions for individual sources. Thus, the section 126 administrative proceeding may be the only way for states to remedy pollution caused by the cumulation of emissions from all stationary sources regulated by a SIP. States have pursued both means of remedy, filing suits and petitions in tandem. Actions have been brought in the context of petitions to review pending and/or approved SIP revisions for individual plants. This activity has culminated in a petition before EPA involving the States of New York, Pennsylvania and Maine -- a consolidation of several section 126 petitions. It requests that EPA require broad-scale reductions in emissions of SO_2 and TSP in midwestern states. Appeals to an EPA determination on the New York-Pennsylvania-Maine section 126 petition might come before the D. C. Circuit Court. Thus, in the absence of Congressional direction to EPA and the courts on long-range transport, the D. C. Circuit Court could ultimately be the arbiter of the long-range transport pollution controversy between the midwestern and northeastern states.

II. Regulatory Mechanisms for Interstate Pollution Control: National Ambient Air Quality Standards and the SIP Process

National Ambient Air Quality Standards (NAAQS) constitute the centerpiece of the environmental protection strategy of the Clean Air Act. Under section 108, EPA prepares a list of "criteria pollutants" which "endanger public health or welfare". Section 109 of the Act requires the Administrator of EPA to adopt primary and secondary ambient air quality standards for these pollutants. Primary standards must be set at levels adequate to protect the public health. Secondary standards, on the other hand, are to be set at levels which will protect the public welfare from "any known or anticipated adverse effects associated with the presence of such pollutant in the ambient air".

While EPA establishes the NAAQS, the primary burden for implementation -- and hence the protection of health and welfare -- is placed on the states by section 110 of the Act. Each state must submit to the Administrator implementation plans designed to attain and maintain NAAQS within its borders. In addition, the SIP must contain:

> Adequate provisions (i) prohibiting any stationary source within the State from emitting any air pollution in amounts which will (a) prevent attainment or maintenance by any other State of any such national primary or secondary ambient air quality standard, or (b) interfere with measures required to be included in the applicable implementation plan for any other State under Part C to prevent significant deterioration of air quality or to protect visibility and, (ii) insuring compliance with requirements of section 126, relating to interstate pollution abatement. (Section 110(a)(2)(E).

The Administrator may not approve a SIP or a SIP revision that does not meet these requirements. Moreover, the Administrator can require revision of a SIP on the basis that it fails to comply with section 110(a)(2)(E). Thus, the statute requires the Administrator and EPA to ensure that states adopt and implement measures to control interstate pollution.

The Clean Air Act provides two means of a remedy for those who believe that current SIPs or proposed SIP revisions fail to adequately control interstate pollution. The Act provides for administrative review of an alleged interstate pollution problem. States and political subdivisons may petition the EPA for a finding of pollution in violation of section 110(a)(2)(E). The Act also provides for judicial review of the Administrator's actions under two provisions of the Act. Litigants may use these provisions as a means to obtain review of the Administrator's actions with regard to controlling interstate pollution through the SIP process. Those eligible to file suit under this procedure include states, industries, environmental organizations and individuals.

III. Utility of the Clean Air Act for the Control of Long Range Transport Air Pollutants (LRTAPs)

The utility of the Clean Air Act for the control of interstate pollution is determined by two factors: 1) The efficacy of the SIP process, in general, for the control of pollution caused by LRTAPs, and 2) the efficacy of the interstate control requirements for SIPs as established by section 110(a)(2)(E). Although the statute leaves the Administrator much latitude to develop interstate control strategies based on the SIP process, the SIP process itself and the language of section 110(a)(2)(E) limit the efficacy of the statute for the control of pollution caused by LRTAPs.

A. Limitations of the NAAQS/SIP Process

1. Focus on the Control of Ambient Air Quality Within the Locale of Emission

The primary focus of the SIP process has been to protect local air quality within a state and to accomplish this goal through the control of individual sources. In practice, states establish emission limitations for sources through the use of mathematical models. The models and air quality monitors are used to set limitations on individual sources to ensure that ground-level air pollution concentrations within 50 km of sources do not exceed NAAQS. However, ground level concentrations close to a source may not reflect the concentrations of LRTAPs, such as sulfates or ozone, which can form high above the ground from the cumulation and transformation of emissions from many sources. Evidence suggests that LRTAPs may remain aloft for several days and be transported over distances far exceeding 50 km to contribute to pollution over broad regions. Hence the SIP process, as currently administered to protect local air quality, does not adequately account for pollutants transported long distances to areas outside the state of origin.

2. Secondary Standards and Welfare Effects

Implementation of the Clean Air Act to date has concentrated primarily upon the mitigation of pollution-related health effects through the health-based primary standards. While health effects associated with long-range transport air pollutants are a topic of great controversy, "welfare effects," such as impacts on materials, agriculture, forests, and aquatic ecosystems, are of foremost concern with respect to interstate and transboundary pollution caused by LRTAPs. However, unlike the primary standards which the Act requires must be achieved within three years of EPA approval of SIPs, the secondary standards are to be attained within a "reasonable time." This general guideline for implementation does not encourage the attainment and enforcement of secondary standards.

Moreover, the NAAQS approach deals only with the concentration of pollutants in air, not the deposition of pollution. Potential interstate or transboundary damage from LRTAPs may be more closely related to the amount of pollutants reaching the ground than to the concentration of these same chemicals or their precursors in air. Ambient concentrations do not necessarily directly correlate with total deposition in an area, since deposition is a function of the intensity and frequency of rainstorms as well as of pollutant concentrations. Thus NAAQS may not adequately address potential damage caused by deposition.

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B. Limitations of Section 110(a)(2)(E)

Section 110(a)(2)(E) prohibits certain interstate air pollution caused by stationary sources and it applies only to pollutants for which there are NAAQS. At present no NAAQS exist for pollutants which are the major cause of acidic deposition -- sulfates and nitrates. Sulfates and nitrates for the most part are not directly emitted into the atmosphere. Rather, they are particulates formed from the chemical transformation of directly-emitted pollutants -- sulfur dioxide and nitrogen oxides. While specific ambient air standards exist for these directly-emitted pollutants -- and for total suspended particulates, of which sulfates and nitrates are a fraction -- no such standards exist for the transformation products themselves. Under the Act, however, the Administrator does have the latitude to adopt NAAQS for pollutants formed through transformation so that standards could potentially be established for sulfates and nitrates. [4]

Other difficulties with the provision limit its utility to the states and Administrator as a means to control interstate pollution caused by LRTAPs. First, the section does not control mobile sources emissions which contribute to the production of ozone and acidic substances that may be transported over long distances. Second, the section gives the Administrator little guidance as to the amount of interstate pollution prohibited by the Act. It appears to establish two different standards by prohibiting pollution that 1) prevents attainment and maintenance of NAAQS, on the one hand, and 2) interferes with measures required to prevent significant deterioration and to protect visibility on the other. Neither the section nor the legislative history offer further explanation of the two terms.

The section also does not specifically require that states consider the long-range effects of the aggregate emissions of a number of sources. Moreover, it does not outline the level of proof of causation which is required to substantiate regulation pursuant to section 110(a)(2)(E). This gives rise to such questions as whether pollution must be linked to individual sources to justify control, or whether regulation of individual sources may be justified through the use of models establishing relationships between the emissions and air quality of "source" and "receptor" regions respectively. This latter point is of significance in the context of LRTAPs because there are no models to identify the contribution of an individual source far downwind, but States in the northeast have argued that models are available to establish cause and effect relationships among regions.

C. Section 110(a)(2)(E) as Applied by EPA

The EPA has reviewed and approved a number of revisions to state implementation plans since the enactment of the section 110(a)(2)(E) requirement in 1977.

Although the Agency has been faced with a number of emission relaxation decisions, the agency has not adopted regulations pursuant to section 110(a)(2)(E) to spell out how states may be sure that proposed revisions comply with the interstate control requirements of section 110(a)(2)(E). The Agency has not reviewed any SIP as a whole to determine whether it complies with the interstate requirements. Nor has it articulated definitive policies for determining whether an individual source relaxation did not violate section 110(a)(2)(E). Current agency practices, however, limit the scope of the SIP review to that portion undergoing revision. Hence, where SIP emission limitation relaxations for individual plants are proposed, EPA considers only the air quality impact resulting from the change in emissions for the named plants. The Agency also takes the position that there are no adequate modeling tools to 1) assess the long-range impacts of individual or multiple sources, or 2) consider the potential impacts of the transformation products of the pollutants for which a SIP relaxation is being considered. (For a detailed description of EPA procedures see Attachment A.)

IV. Administrative and Judicial Remedies for Interstate Pollution under the Clean Air Act

A. Judicial Review as a Means to Remedy Interstate Pollution Caused by LRTAP

Two sections of the Act provide for judicial review of actions taken by the Administrator under the Clean Air Act. Section 304 provides that any person may commence a civil action on his own behalf in U.S. District Court against the Administrator for a failure to carry out a duty made mandatory by the Act. Section 307(b)(1) provides that a petition for review of actions specified in the section can be brought in the U.S. Circuit Courts.

Petitioners have filed suit pursuant to section 307(b)(1) to challenge the Administrator's approval of SIP relaxations on the basis that both the relaxations and the SIPs as a whole violate the requirements of section 110(a)(2)(E). Petitioners have also argued in the context of section 307 suits that the Administrator has a mandatory duty to review each state's SIP, as a whole, for its compliance with section 110(a)(2)(E), regardless of whether the state has proposed a revision to its SIP subsequent to the enactment of section 110(a)(2)(E) in 1977.[5] Since EPA contends that there is no mandatory duty to perform such a review, it has not done so to date. No suits have been filed pursuant to section 304, concerning interstate pollution, to compel the Administrator to review SIPS as a whole even in the absence of proposed revisions to a plan. This may suggest that litigants are not optimistic about the chances that a court would accept the mandatory duty argument and take jurisdiction under section 304. However, in the absence of a court ruling to the contrary, section 304 remains available as a potential means to compel the Administrator to take action on LRTAPs, on the basis that such action is made mandatory by the Clean Air Act.

B. Section 126: The Administrative Petition Process

Section 126 is a companion section to section 110(a)(2)(E). It requires states to provide notice to nearby states of sources which "may significantly contribute" to air pollution in excess of NAAQS. In addition, it provides an administrative means to remedy interstate pollution through petition to the Administrator of EPA. Section 126(b) provides:

> Any State or political subdivision may petition the Administrator for a finding that any major source emits or would emit any air pollutants in violation of the prohibition of section 110(a)(2)(E)(i). Within 60 days after receipt of any petition under this subsection and after public hearing, the Administration shall make such a finding or deny the petition.

If the Administrator makes a finding of violation under section 126(b), then it is a violation of SIP for a proposed source to be constructed or to operate in violation of section 110(a)(2)(E) and for an existing source to operate more than three months after the date of the finding. However, the Administrator may permit continued operation beyond three months if a source submits to a compliance schedule. Like section 110(a)(2)(E), section 126 only addresses pollution from stationary sources and it does not provide the direct means to control non-criteria air pollutants or pollutant deposition; it is limited to the direct control of air pollution caused by NAAQS pollutants.

Neither section 126 nor the legislative history provide guidance on the procedures the Administrator should follow with regard to the statutorily-required hearing. Similarly, section 126 offers no more guidance than section 110(a)(2)([illegible]) as to what quantum of interstate pollution is prohibited under the Act -- the standards of section 110(a)(2)(E) are those to be used in a 126 proceeding.

C. Comparison of the Section 126 Petition Process and Judicial Review of SIP Approvals

The section 126 petition process and the judicial review process under section 307(b) and section 110(a)(2)(E) differ in terms of: who may seek remedy, and what remedies they may seek under the two review procedures. The section 126 petition process is reserved for states and political subdivisions; the section 307(b) review of SIP approvals is not limited to a particular class of litigants. A recent court decision also suggests that section 307(b) review is limited to review of the particular SIP revision; thus, litigants cannot seek remedy for the emissions of all sources regulated by the SIP in the context of a SIP revision for individual plants. It appears that the section 126 petition process, on the other hand, is available as a means to remedy regional pollution problems. Both sections 126 and 307(b) are limited in scope to air pollution caused by NAAQS pollutants.

D. Section 307 Suits Involving the LRTAP Issue

States and other litigants seeking remedy for interstate pollution caused by LRTAP, have challenged, pursuant to section 307, EPA approval of SIP revisions for single sources, and in the course of these suits have attempted to obtain judicial review of EPA's refusal to address regional emissions through the SIP process. They have also attempted to compel EPA to use models and other techniques to assess long-range transport effects. Litigants have argued that the approval for the SIP revision is in error for two reasons: 1) the Administrator failed to properly review the LRTAP effects of the individual source for which a relaxation was approved because EPA relies only on "approved models" to estimate impacts within 50 km of the source, 2) the Administrator failed to review the whole SIP to be sure the cumulative emissions of sources governed by the SIP does not cause pollution in violation of section 110(a)(2)(E).

Suits have been filed in the D.C., Second, Third, Sixth, and Seventh Circuits by states and environmental groups. No court has yet directly ruled on the issue of whether EPA must use the best available techniques to assess long-range transport effects. Two cases have been decided that indicate that the petitioners may not be able to get review of the underlying SIP or to compel EPA to review SIPs in general for their adequacy in controlling LRTAP.

In Connecticut v. EPA, the Second Circuit held that the proper scope of review under section 110(a)(2)(E) was of the emissions relaxations proposed for individual sources, and the impact of multiple sources governed by the SIP could not come before the court in the context of a challenge to a section 110(a)(2)(E) SIP revision. In a subsequent case, New York State challenged

EPA's approval of an Ohio SIP revision in the D.C. Circuit on the basis that EPA's approval was unlawful since it did not review the SIP as a whole; moreover, New York claimed that EPA had a duty to review all SIPs for their compliance with section 110(a)(2)(E). New York argued that the D.C. Circuit had jurisdiction over the issue because the issues presented were of nationwide scope and effect -- a basis for jurisdiction in the D.C. Circuit. The court, however, never addressed the merits of New York's claims since the court dismissed the case without opinion for lack of jurisdiction. The cases were refiled in the Sixth Circuit, where they are still pending.[6]

1. The Judicial Review of EPA Policies for Modeling and Proof of Causation

The cases which are pending before the courts challenge the EPA's alleged failure to consider the LRTAP of single sources as well as its refusal to review underlying SIPs in the context of emission relaxations for specific sources. Challenges have been filed, among others, by the States of Pennsylvania, New York, Connecticut and Massachusetts in the suits pending before the Seventh, Sixth and Second Circuits.[7]

States have used the section 307 challenges to SIP revisions for individual plants as a means to challenge EPA's policies with regard to modeling and proof of interstate impact. Massachusetts, Pennsylvania, New York and New Hampshire with the Northern Ohio Lung Association, for example, have challenged EPA approval of SIP relaxations for two plants in Ohio, in part, on the basis that the Administrator's decision is in error because EPA did not review the long-range transport impacts of the primary pollutant SO_2 or the secondary pollutant -- sulfates.[8] In addition, they did not consider the long-range transport effects of the total emissions governed by the Ohio SIP. They reject the EPA position that there are no models appropriate for regulatory purposes that can estimate impacts beyond 50 km. The petitioners point to comments made during the rule-making which allegedly demonstrate the existence of analytical tools to link emissions of regions to air quality downwind and demonstrate the existence of long-range transport models to predict the impact of multiple source emissions. 9/ They argue EPA has a duty to use the best available of these techniques to consider long-range transport effects. The petitioners argue that scientific uncertainty does not make EPA's position legally defensible since "Congress did not intend EPA to withhold action on the interstate pollution problem while awaiting the 'Godot of scientific certainty'". 10/

Petitioners have also challenged the EPA position that it has no duty to consider the air quality impacts on particular levels of the transformation products of SO_2 when faced with a proposed SO_2 relaxation. They argue that: 1) there are NAAQS for particulates; 2) SO_2 is transformed to sulfates which are particulates; and that therefore, 3) SO_2 emissions must be controlled to ensure they don't interfere with PSD increments for TSP or prevent attainment and maintenance for NAAQS. 11/ Petitioners claim that EPA's position "(flies) in the face of the plain language of the Act, which does not exclude regulation of precursors", moreover, they note that EPA regulates precursors in the case of ozone, a criteria pollutant.

No court has directly ruled on the issues presented by the petitioners; that is, whether EPA must assess long-range transport effects of individual or multiple sources in spite of the fact there are no approved models appropriate for such assessments and secondly, if it must, whether EPA must use best available modeling techniques which may only be capable of establishing linkage between the emissions and air quality of regions. Thus, it is an open-ended question whether proof of the impact of emissions from a region is sufficient proof to require regulation of individual sources under section 110(a)(2)(E). In addition, it remains an open question whether EPA must assess the air quality effects of SO_2 emissions on particulate concentrations.

E. Activity Under Section 126

Petitions pursuant to section 126 have been filed by Jefferson County, Kentucky, and the States of New York, Pennsylvania, Connecticut, Maine, New Jersey, Kentucky and West Virginia. Petitioners have sought remedy for pollution caused by single sources within a short distance of emissions, and redress for pollution caused by long-range transport of emissions from individual and multiple sources.

Section 126 petitions involve EPA in highly complex analytical procedures; however, the agency is statutorily required to reach determinations on them within 60 days. Although a number of petitions have been filed since 1977, to data a determination has been made on only one of them -- the Jefferson County, Kentucky petition concerning alleged pollution caused by the Gallagher power plant situated just across the Indiana state border. EPA held a public hearing on the petition in April 1980, and issued a determination denying the petition on February 16, 1982.

Agency enforcement actions taken subsequent to a section 126 petition filed and later withdrawn by the State of West Virginia suggest that the section is reasonably effective in situations where interstate pollution is caused by sources close to the border and where petitioners may rely on agency approved models to establish causation. It is more difficult to assess its effectiveness as a means of remedying pollution caused by long-range transport. Direct administrative remedy under section 126 may be foreclosed by EPA since it contends that there are no tools to estimate impacts caused by long-range transport. Remedy could also be foreclosed if EPA interprets section 126 as placing a heavy burden of proof of interstate pollution on petitioners and requires that emissions be traced to sources.

Ultimately, the D.C. Circuit Court could be faced with an appeal to a section 126 determination dealing with the long-range transport controversy. The appeal could require the Circuit to address the fundamental question of whether EPA must use best available modeling techniques for long-range transport and develop a consistent policy to deal with regional air quality in the eastern U.S. In short, in the absence of explicit Congressional guidance on pollution caused by long-range transport, the long-range transport controversy may initially be determined by the D.C. Circuit Court.

1. Burden of Proof Under Section 126

In the process of making a determination on the Gallagher petition, the EPA Administrator announced criteria which set forth information that EPA considers necessary to trigger consideration of an alleged problem under section 126, 46 F.R. 3839, (July 30, 1981). The final determination, however, explicitly stated that the criteria were only guidelines.[12] One can glean some insight, however, as to what the EPA believes the burden of proof should be under section 126 from the guidelines and EPA responses to comments concerning them. In brief, it appears that section 126 petitioners need only make out a prima facie case, and then the burden shifts to EPA to support the determination.

EPA noted in the final determination regarding the burden of proof, that in at least initial stages the petitioners have the burden of coming forward with information as the proponent of agency action and as those with access to information forming the basis of the petition. 13/ EPA had requested the following information from petitioners:

> (1) A demonstration establishing the existence and geographic boundaries of the nonattainment or PSD area which is the subject of the petition;
>
> (2) A demonstration that achievement of ambient air quality standards or of measures necessary to prevent significant deterioration or to protect visibility, is prevented by the named out-of-state sources;
>
> (3) An indication that sources within the petitioning State(s) which effect PSD and nonattainment areas have been adequately controlled. 46 FR 38939 (July 30, 1981).

EPA goes on to note, however that section 307(d) 14/ of the Act requires that EPA follow informal rule-making procedures in reaching a final determination under section 126. This includes, in part, a duty "to compile factual data in support of its ultimate determination," in a statement of basis and purpose. 47 FR 6626 (February 16, 1982). Thus, EPA acknowledges that it ultimately has the burden of proof in establishing justification for the final determination. 15/

The Gallagher determination also touches on EPA's interpretation of the first part of the section 110(a)(2)(E) prohibition -- i.e., what constitutes pollution that prevents attainment and maintenance of NAAQS. In the Gallagher case, a 3 percent contribution to a violation was "not sufficiently significant to support an affirmative finding." 47 FR 6626 (February 16, 1982).

The determination also observes that:

> The Clean Air Act itself provides no guidance on the levels of pollution which constitute prevention of attainment and maintenance of the NAAQS. However, the legislative history of section 126 suggests that a strict literal interpretation of the term "prevent" is not intended. Because a state is usually able to set tighter emission limits on its own sources to offset the effects of interstate pollution, a literal interpretation of the term "prevent" would rarely allow relief under section 126. This interpretation would fail to address interstate inequities that Congress intended section 126 to rectify. See S. Rep. No. 95-127 at 41-42 and H.R. Rep. No. 95-294, 95th Cong., 1st Sess., May 12, 1977.

However, the Gallagher determination does not establish an EPA position on the amount of pollution prohibited under section 110(a)(2)(E), and offers little information on how EPA will rule on the section 126 petitions.

2. Petitions under Section 126 Seeking Remedy for LRTAPs

States seeking remedy of interstate pollution caused by the long-range transport of emissions from multiple sources have filed section 126 petitions in tandem with suits under section 307. The petitions of the States of New York, Pennsylvania and Maine have been consolidated into a single proceeding, which places before EPA the gamut of issues involved in the long-range transport controversy. Petitioners have asked for far-reaching relief for the interstate pollution problems they allege, including revision of EPA policies and broad-scale reductions of emissions in the eastern United States.

EPA appears to have allowed that aggregate impacts can be at least considered in a section 126 petition. In Pennsylvania v. EPA, the agency argued before the Third Circuit Court that a section 126 proceeding, rather than a section 110 proceeding, was the appropriate forum for considering the efficacy of the one underlying SIP to control for interstate pollution caused by the aggregate emissions of all SIP-governed sources. 16/

Any determination reached by EPA will probably be challenged in the courts. Due to the "national nature" of the controversy, it appears that a petition for review of an EPA determination might be filed in the D.C. Circuit Court of Appeals. Thus, the D. C. Circuit could be forced to grapple with the difficult issues presented by the section 126 proceeding when assessing the rationality and procedural sufficiency of an EPA determination. 17/

The standard of review that the Court would apply in such a case is set out in section 307(d)(9). Generally, the section provides that the Court may reverse any action of the Administrator found to be arbitrary and capricious, an abuse of discretion, not in accordance with law, or without observance of procedure required by law. 18/ Judicial review is limited to issues raised in the administrative proceedings before EPA, and would focus on EPA's statutorily-required statement of basis and purpose for the proposed rule-making. 19/

a. Issues Raised in the New York-Pennsylvania-Maine Section 126 Proceeding

The consolidated petitions claim that SO_2 and TSP emissions of sources governed by SIPs in the states of Illinois, Indiana, Kentucky, Michigan, Ohio, West Virginia and Tennessee are causing interstate pollution in violation of section 110(a)(2)(E). The petitioners allege that such pollution prevents attainment and maintenance of NAAQS for both TSP and SO_2 and interfere with PSD requirements. Petitioners further allege that the conversion of SO_2 to particulate sulfates contribute to the aforementioned problems; in addition sulfates present a hazard to health and cause acid rain and, as a result, cause damage to aquatic and terrestrial ecosystems. Maine specifically claims that conversion of SO_2 emissions causes visibility degradation in the mandatory Class I area of Acadia National Park in violation of the Act. Petitioners have asked that EPA review the SIPs for long-range transport effects and have requested major reductions in emissions in the seven midwestern states named in the petition. In developing the arguments to support their petition, petitioners have touched on a number of issues: 1) the burden of proof under section 126; 2) the amount of pollution prohibited by section 110(a)(2)(E); 3) the duties of EPA under the interstate provisions; and 4) whether EPA must require the regional emissions reductions requested by the petitioners.

Petitioners have presented extensive information on: the air quality problems and alleged environmental problems resulting from acid rain in their states; air quality data and modeling results which allegedly demonstrate that interstate pollution contributes significantly to these problems; air quality models and analytical techniques available to assess the long-range transport of air pollutants; the relative stringency of emission controls between the midwestern states and the petitioning states and, finally, the alleged inequities arising between the midwestern states named in the petition and the states of New York, Pennsylvania and Maine as a result of the different levels of emission control which petitioners claim are needed to control interstate pollution.

Extensive comments have been filed utilities, environmental groups, and other intrested parties concerning the information presented by the petitioners and the issues raised. As yet, EPA has issued no preliminary findings in the proceeding.

F. Relief Possible in Challenges to SIP Approvals Under Section 307 and in Section 126 Proceedings

It is not clear what relief the petitioners could win if successful in the suits pending before the courts or in the section 126 proceedings before the agency. However, a wide range of results are possible. Neither route is a means to compel direct control of acidic deposition. Litigants under section 307 seeking EPA consideration of long-range transport effects could achieve a court decision which requires such a review with "state-of-the-art techniques." The scope of the New York-Pennsylvania-Maine section 126 petitions is wider; the petitioners could, if successful, win a reversal of EPA SIP review policies, and EPA could require a major reduction in emissions in the Eastern United States. EPA could agree, or the courts could rule, that petitioners had presented sufficient evidence to justify the assertion that regional emissions are causing impermissible interstate effects, and require revisions to all applicable SIPs on that basis. Any state-by-state allocation formula devised by EPA for emissions reductions would be subject to judicial scrutiny.

FOOTNOTES

1/ Section 102(c) of the Act explicitly notes another means which states, in theory, could use to augment the requirements of the SIP process to control insterstate pollution caused by LRTAPs. Section 102(c) provides that states may enter into agreements or compacts to control air pollution and such agreements are binding if approved by Congress.

Interstate compacts could, in theory, be used to address interstate pollution caused by LRTAPs. In practice, the interstate compact approach involves difficulties which make it an unlikely means to resolve interestate pollution caused by LRTAPs. First, compacts typically have taken years to negotiate. Second, the regulatory agencies established under them have been inadequately funded and oftentimes have not been granted enforcement authority. Third, in the context of a LRTAP pollution problem, states which "export" more polllution than "import" have little incentive to voluntarily enter into compacts which might require emission reductions to control LRTAPs. To date no interstate compacts or agreements have been established pursuant to Section 102(c) of the Clean Air Act. See, e.g., Curlin, "The Interstate Water Pollution Compact: Paper Tiger or Effective Regulatory Device", 2 Ecology L.Q. 333 (1972) and Dixon, "Constitutional Bases for Regionalism and Centralization, Interstate Compacts; Federal Regional Taxation" 33 Geo. Wash. L. Rev. 47 (1964) for a discussion of the efficacy of interstate compacts.

2/ 42 USC section 7401 et seq. (1980).

3/ The statement of Douglas Blazey, Chief Counsel, Pennsylvania Department of Environmental Resources before the Senate Committee on Environment and Public Works is illustrative of the position of some states:

> Since 1977, the Commonwealth, through the department, has attempted to have the U.S. EPA implement the interstate pollution abatement provisions of the Clean Air Act.
>
> To date, EPA has not, it can fairly be said, taken any affirmative action to reduce interstate air pollution and has never taken a long-range transport phenomenon into account when making regulatory decisions...
>
> It justifies its actions by stating that it has conducted research concerning the long-range transport of air pollutants, and that the state-of-the-art dispersion modeling does not enable EPA to adequately assess long-range impacts in its regulatory decisions.
>
> While EPA may genuinely believe that one cannot precisely quantify the contributions of a single source of SO_2 emissions in one State to the SO_2 sulfate and acid rain problems in another State, this view does not negate the fact that SO_2 emissions in one State individually and collectively are being transported long distances downwind, affectng air quality in other States.

Hearings Clean Air Act Oversight, Senate Committee on Environment and Public Works, June 30, 1981. No. 97-H12, pp. 259-260.

4/ Ozone is an example of a pollutant formed through chemical transformation of directly-emitted pollutants. There are NAAQS for ozone itself, as well as for its precursors nitrogen dioxide and hydrocarbons. (The hydrocarbon standard is for use as a guide in attaining the ozone standard.) Ozone and its precursors can be transported long distances.

5/ For example, New York is arguing before the 6th Circuit that EPA violated the Act in approving revisions to the Ohio and Michigan SIPs by failing to assess the LRT impacts of the revisiosn on TSP and sulfate levels in New York. New York v. EPA 6th Circuit Court #81-3357. Moreover, it asserts that the Administrator has a mandatory duty under the law to consider the impacts not only of the plants for which revisions are proposed, but also to review the complete Ohio and Michigan SIPs for cumulative effects of all sources regulated byi them. New York claims such a duty is imposed by section 406(d)(2). Section 406(d)(2) provides:

> "Except as otherwise expressly provided, each State required to revise its applicable implementatin plan by reason of any amendment made by this Act shall adopt and submit to the Administrator of the Environmental Protection Agency such plan revisions before the later of the dates --
>
> (A) one year after the date of enactment of this Act, or
>
> (B) nine moNths after the date of promulgation by the Administrator of the Environmental Protection Agency of any regulations under an amendment made by this Act which are necessary for the approval of such plan revision."

New York has alleged that, since section 10(a)(2)(E) was adopted by amendment in 1977, and because the SIPs have not undergone a review for their sufficiency with respect to 110(a)(2)(e), then the Administrator has a duty to review the SIP as a whole and revise it, as necessary, to bring it into compliance with section 110(a)(2)(E).

6/ New York petitioned the D. C. Circuit for review of the Administrator's approval of Ohio and Michigan SIP revisions, arguing that the Court had jurisdiction over the issue because:

> "the issues (were) of nationwide scope and effect since they involve a uniform failure of the Administrator to implement, enforce and comply with critical provisions of the CAA".

Customarily a litigant would file a petition in the Circuit in which the State revising its SIP is located, because section 307(b)(1) of the Act provides, in pertinent part, that:

> A petition for review of the Administrator's action in approving or promulgating an implementation plan under Section

> 110 or Section 111(d) ... or any other final action of the Administrator under this Act ... which is locally or regionally applicable may be filed only in the United States Court of Appeals for the apropriate circuit. 42 U.S.C.A. § 7607(b)(1) (Supp. 1979). (Emphasis added.)

The D. C. Circuit dismissed the New York petition without opinion for lack of jurisdiction. Thus the court did not address the merits of New York claims. The cases were refiled in the Sixth Circuit. New York v. EPA, 6th Cir. 81-3357 and 81-3356.

7/ See, for example, Connecticut Fund for the Environment v. EPA 2nd Cir. #81-4228 (challenge to New York SIP revision); Pennsylvania v. EPA, 6th Cir. #81-3304 (challenge to Ohio SIP revision); New York v. EPA 6th Cir. #81-3357 (challenge to Michigan SIP revision); Ohio et al. v. EPA 6th Cir. #80-3575 (Challenge to Ohio SIP revision); New York v. Gorsuch, 6th Cir. 82-3126 (challenge to Tennessee SIP revision); New York v. Gorsuch, 6th Cir. 82-3426 (challenge to Ohio SIP revision); New York v. EPA, 7th Cir. 80-2808 and 82-1418 (challenge to Illinois SIP revision); New York v. Gorsuch, 7th Cir. 82-1717 (challenge to Indiana SIP revision); New York v. Gorsuch, et al., 7th Cir. 82-2059 (challenge to Indiana SIP revision).

8/ Ohio et al. v. EPA, 6th Cir. 80-3575 (consolidated), Brief for Petitioners Commonwealth of Massachusetts, et al.

9/ Id. at 38-54.

10/ Id. at p. 49.

11/ Id. at 37-38.

12/ The final determination states:

> ... the criteria are only intended to guide petitioners in providing information that is useful in a review of a section 126 petition. They are by no means exhaustive nor do they represent requirements or prerequisites for an affirmative determination under section 126. ...
>
> ... the Administrator considers it inappropriate at this time to issue regulations which would constitute a rigid formula for section 126 determinations. 47 F.R. 6625 February 16, 1982.

13/ 47 FR 6626 (February 16, 1982).

14/ Section 307(d) prescribes rulemaking procedures for certain actions of the Administrator including action under section 126. The submission requires that EPA publish a notice of proposed rulemaking in the Federal register which shall include a statment of basis and purpose for the proposed rules. The EPA is also required to establish a docket and interested parties may submit comments on the proposed rule and place them in the docket. In the promulgating the rule the Administrator is required to allow written comments as well.

15/ The significance of the "informal criteria" established by EPA for section 126 petitions dealing with long range transport is not clear. A strict interpretation of criterion #2 could make it difficult for a petitioner to establish a claim sufficient for consideration under section 126. Since EPA has approved no models to assess the long-range transport effects of individual sources, EPA might not accept a demonstration based on other techniques. Indeed, EPA has argued before the courts in the section 307 suits that it cannot address the long-range transport issue because available tools lack sufficient accuracy. Secondly, the criteria focus on identifying the impact of specific sources, and this is more difficult, if not impossible, in the context of long range transport.

The third criterion deals not so much with burden of proof, but rather with issues of equity. Information on the level of control in the petitioning state can be used by EPA to frame a determination that considers the relative stringency of regulation between the petitioning state or poltical subdivision and that which allegedly is "exporting" pollution in violation of section 110(a)(2)(E).

16/ EPA noted in its brief with regard to the New York-Pennsylvania-Maine section 126 proceeding that "[the proceeding] will be the appropriate forum for evaluating Pennsylvania's claims regarding cumulative out-of-state source impacts. In contrast to the source-specific review required under section 110(a)(2)(E) for individual SIP revisions, such as those challenged here, emissions impacts due to any identified out-of-state sources can be considered in a section 126 proceeding." Brief for Respondents at 35, Pennsylvania v. EPA, 3rd Cir. # 79-1025.

17/ Section 307(b)(1) provides in relevant part:

> A petition for review of action of the Administrator in promulgating any national primary or secondary ambient air quality standard ... or any other nationally applicable regulations promulgated or final action taken by the Administrator under this Act may be filed only in the United States Court of Appeals for the District of Columbia. . . . a petition for review of any action . . . may be filed only in the United States Court of Appeals for the District of Columbia if such action is based on a determination of nationwide scope or effect and if in taking such action the Administrator finds and publishes that such action is based on such a determination.

Hence, jurisdiction could lie in the D.C. Circuit for at least two reasons. First, due to the national nature of the controversy, the Administrator of EPA might find that a determination made in the New York-Pennsylvania-Maine section 126 proceeding had "nationwide scope or effect." Section 307(b)(1) provides that the D.C. Circuit has exclusive jurisdiction over such matters. Second, petitioners or EPA could argue that a determination made in the New

York-Pennsylvania-Maine section 126 proceeding was a "nationally applicable" final action taken by the Administrator and, as such, section 307(b)(1) also requires that a challenge be brought only in the D.C. Circuit.

In the absence of an Administrator's finding of "nationwide scope or effect," EPA could be faced with defending lawsuits in a number of different circuits. Section 307(b)(1) provides that challenges to final actions which are regionally applicable must be brought in the "appropriate circuit". Since the New York-Pennsylvania-Maine petition involves seven midwestern state SIPs and alleges impacts in the three petitioning states, without a determination of nationwide scope and effect, petitions for review arguably could be filed in at least six circuits.

18/ The full text of section 307(d)(9) provides:

> (9) In the case of review of any action of the Administrator to which this subsection applies, the court may reverse any such action found to be --
>
> (A) not in accordance with law;
>
> (B) contrary to constitutional right, power, privilege or immunity;
>
> (C) in excess of statutory jurisdiction, authority, or limitations, or short of statutory right; or
>
> (D) without observance of procedure required by law, if (i) such failure to observe such procedure is arbitrary or capricious, (ii) the requirement of paragraph (7)(B) has been met, and (iii) the condition of the last sentence of paragraph (8) is met.

Section 307(d)(7)(B) provides generally that only an objection to a rule or procedure which was raised during the period for public comment can be raised during judicial review. Section 307(d)(8) provides, in relevant part, that a court may invalidate a procedural error only if the erorrs were so serious "there there is a substantial likelihood that the rule would have been significantly changed if such errors had not been made."

19/ Section 307(d)(3) provides, in relevant part,

> " . . . The statement of basis and purpose shall include a summary of--
>
> (A) the factual data on which the proposed rule is based;
>
> (B) the methodology used in obtaining the data and in analyzing the data; and
>
> (C) the major legal interpretations and policy considerations underlying the proposed rule.

The statement shall also set forth or summarize and provide a reference to any pertinent findings, recommendations, and comments by the Scientific Review Committee established under section 109(d) and the National Academy of Sciences, and if the proposal differs in any important respect from any of these recommendations, an explanation of the reasons for such differences. All data, information, and documents referred to in this paragraph on which the proposed rule relies shall be included in the docket on the data of publication of the proposed rule.

ATTACHMENT A

Methodology Used by EPA to Review Proposed SIP Revisions

To determine whether a proposed emission limitation for a source will cause violations of the Act, the EPA uses air quality data and mathematical models to establish a "background level" of air pollution. "Background" is ambient concentration of pollution that exists in the absence of the emissions from the source for which the revision is proposed. EPA estimates the air pollutant concentratons caused by the source under the conditions of the new emission limitation. Estimates are derived using mathematical models approved by EPA which are suitable to estimate pollutant concentrations for the particular case. EPA, in theory, then takes the results of such modeling studies and determines whether the proposed emission limitation will ensure compliance with requirements of the act. That is, the agency presumably uses the results to determine whether a revision will prevent attainment or maintenance of NAAQS or interfere with PSD requirements in a downwind state. As shown below, however, the EPA's position on acceptable modeling techniques precludes consideration of long-range transport effects through the use of models.

First, in reviewing the emission limitations with respect to the interstate provisions the agency only considers impacts which can be estimated by models approved pursuant to an agency modeling guideline. This effectively limits consideration of impacts to within 50 km of source since EPA has approved no models to estimate long-range transport impacts which could be felt well beyond this distance. EPA does not consider accurate enough for regulatory purposes those models that could presently be used to estimate long-range transport impacts. Second, the agency does not consider potential impacts of the transformation products of the pollutant for which a SIP relaxation is being considered. The Agency reviews the interstate effect of an SO_2 relaxation, for example, solely for its impacts on SO_2 concentrations in downwind states. It does not consider its impact on TSP levels for which there is a NAAQS and for which there are allowable pollutant increments under the PSD provisions, nor does it consider the effect of the increased SO_2 emissions on visibility. EPA justifies this approach on the basis that there are no approved models that can model the transformation of SO_2 to sulfate -- a fine particulate. Hence, it is not capable of assessing impacts of the transformation products formed during long-range transport. Moreover, EPA contends it is not required by the Clean Air Act to review an SO_2 proposed SIP relaxation for its effects on TSP pollutant concentrations. Thus, EPA does not directly consider long-range transport effects in its review of SIP relaxations.

The above-mentioned methods used to review a proposed SIP revision do not indicate what amount of pollution EPA believes is prohibited under section 110(a)(2)(E). The Agency has made no policy statements clarifying the issue. However, the Agency has held that contribution to a violation of NAAQS will in itself not preclude approval of a SIP relaxation. In the case of a SIP revision proposed to relax emission limits for the Gallagher Plant in Indiana, EPA found that the revision met the requirements of the Act even though it contributed to pollution in violation of NAAQS in the State of Kentucky. The contribution to the total concentration which exceeded the NAAQS was deemed insignificant and, hence, allowable.[*]

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[*] In estimating the impact of a source on a downwind state EPA makes the estimates under meteorological conditions and at sites in the downwind state where the pollutant concentrations are believed to be the highest thereby predicting the "worst-case" contributions to pollution in the downwind state. EPA customarily considers the "worst case" to be highest, second high estimate among the sites for which the model made predictions over the same time period. See 46 FR 37645, July 22, 1981.

EPA uses two other techniques in reviewing SIP relaxations which have implications for the definition of unlawful pollution under section 110(a)(2)(E). Both techniques tend to shift the burden of control to the "downwind" state. First, when EPA determines whether a particular source contribution to pollution is significant to trigger a section 110(a)(2)(E) violation, it customarily does not consider how much of the "background" pollutant concentration is caused by sources in the upwind state. Thus, a state requesting approval for a SIP relaxation could already be contributing a major portion of the "background levels" of pollution in the downwind state. As mentioned earlier, "background" is added to estimates of pollutant impacts of particular sources to establish emission limitations. Therefore, background can have an effect on the stringency of emission limitations prescribed for sources to ensure that NAAQS are attained and protected. It follows that interstate pollution contributions to background can result in stricter limitations in downwind states.

Second, in reviewing a relaxation for its sufficiency with respect to section 110(a)(2)(E), EPA customarily does not consider a source's contribution to pollution levels in areas better than NAAQS for which the PSD baseline has not been triggered. Under the Act the PSD provisions specify that areas with air cleaner than NAAQS must limit pollution increases to specified increments above a baseline level which is defined by statute. Baseline has been construed by the courts as the date when the first PSD application for an area is submitted. 42 USC 7479(4), Alabama Power Co. v. Costle, 636 F2d 323, 374

The estimated "worst-case pollutant concentration" in the Gallagher case was 443 ug/m^3. The plants contribution to this level (exclusive of background) was 15 ug/m^3 or 3% of the total pollutant concentration at the time the standard was violated. 47 FR 6627 February 16, 1982.

(D.C. Cir. 1979). A source's contribution to pollutant concentrations in areas subject to PSD but for which no baseline has been established is not customarily assessed by EPA. Such contributions, although they raise pollutant concentrations in a downwind state, are not usually assessed since they are considered a part of the baseline for the PSD area. It should be noted, however, that the NAAQS operate as a limit on the amount interstate pollution can raise the baseline.

CHAPTER TWO: CONTROL OF TRANSBOUNDARY POLLUTION CAUSED BY LRTAPS: THE U.S.-CANADIAN CONTEXT 1/

I. Introduction

The problems of interstate and transboundary pollution control are strongly linked. Winds that transport pollutants over state lines can also carry them across the border between Canada and the United States. 2/ Hence, long-range transport transforms domestic issues such as air quality enforcement policies and relaxation of emissions limits into issues of international concern. 3/

Mechanisms are available for cooperative solutions to the transboundary issue. There are means to implement control programs and potential means of remedy in the event pollution is inadequately controlled.

The Clean Air Act provides a statutory basis to implement control emissions that may contribute to transboundary air pollution. Moreover, unlike interstate pollution control provisions, the Act applies to transboundary pollutant deposition. In addition, negotiations are proceeding under a U.S.-Canadian Memorandum of Intent (MOI) to develop a bilateral accord. The efficacy of these mechanisms to deal with transboundary pollution depends substantially on the combined commitment of the U.S. and Canada to attempt to control such pollution.

In theory, the Clean Air Act, domestic common law, and international law provide a means of legal recourse should Canada or others be dissatisfied with U.S. control efforts. However, in practice the common and international law may not provide an effective means to compel U.S. control efforts. The Clean

Air Act does not explicitly provide a means of remedy, insofar as the transboundary pollution control provision is activated voluntarily by the Administrator of EPA or the Secretary of State. However, actions taken under the Carter Administration may have created a legal obligation for the EPA to initiate control measures.

Acidic precipitation resulting from the long-range transport of air pollution has emerged as one of the most significant bilateral issues between Canada and the United States. Current U.S. policies are perceived by many Canadians to undercut diplomatic efforts to reach a bilateral accord to control transboundary air pollution. Canadian dissatisfaction with U.S. policies concerning long-range transport has placed strains on U.S.-Canada relations. Most recently, the Canadian Environment Minister, John Roberts, voiced Canadian dissatisfaction with the progress of the negotiations under the MOI, and threatened Canadian withdrawal from them altogether. [4/] However, the Canadian Government might choose to link the transboundary issue to other areas of bilateral concern, in an effort to encourage adoption of policies to reduce the long-range transport of air pollution.

The current Administration has taken the position in the bilateral negotiations that while long-range air pollution in general, and acid deposition in particular, may present serious problems, too many scientific aspects of long-range transport and its effects are inadequately understood to allow us to embark on a bilateral control program. [5/] As the State Department's Canadian affairs officer succinctly states the position: " . . . the scientific relationship between emissions and effects is not clear. We simply don't know where to put the money at the present time to control acid rain." [6/] Hence, while no eventual course of action has been formally barred, the Administration intends to await more information before reaching a decision on bilateral accord.

The Administration also believes that good faith efforts under the MOI do not require EPA to consider the international consequences of emission relaxations under the SIPs -- taking the position that, so long as local NAAQS are protected, the U.S. is vigorously enforcing existing laws as agreed under the MOI. Canadian officials assert that good faith efforts require the U.S. to at least "hold the line" on regional emissions, if not to reduce emissions. Although the status of the MOI is unclear, it does not appear to be legally binding on Canada or on the United States. Its force derives solely from the good intentions of both countries.

A. Recent U.S. Policy

The years 1978 through 1980 were years of intensive cooperative activity with respect to transboundary air pollution and the acidic precipitation issue between Canada and the U.S. In 1978 the United States and Canada established the Bilateral Research Consultation Group (RCG) on the Long Range Transport of Air Pollutants (LRTAP) to assess the transboundary problem. In 1979 the two governments issued the Joint Statement on Transboundary Air Quality. Negotiations culminated in the 1980 Memorandum of Intent between the two governments. The MOI contains strong statements of the two governments' intention to negotiate a treaty on acidic precipitation and LRTAP. In addition, it clearly states that the two governments agree to take interim steps to control emissions causing transboundary pollution.

During the same period, the U.S. took a number of actions indicating that long-range air pollution was viewed as a potentially serious national and international environmental problem. In 1979, President Carter established the Interagency Acid Rain Coordinating Committee and inaugurated a 10-year plan to research the acid rain phenomenon. [7] The Congress sanctioned the President's proposal in 1980 by passing the Acid Precipitation Act of 1980 and authorizing 50 million dollars to support this 10-year acid rain research effort. [8] In 1981, during the closing days of the Carter Administration, the Administrator of EPA made determinations to activate the international air pollution section of the Clean Air Act and to control acidic deposition in Canada. Steps were taken to develop a control strategy.

EPA Administrator Anne Gorsuch has not pursued her predecessor's efforts to address the acid deposition problem through the Clean Air Act. [9] President Reagan has, however, assured Canada that the U.S. commitment to resolve transboundary pollution problems is unchanged. [10] The Administration has proposed no new legislative programs for controlling long-range pollution, acid rain, interstate or international pollution to date. Rather, principles for amending the Clean Air Act announced by EPA Administrator Gorsuch on August 5, 1981 specifically propose a relaxation in the current motor vehicle emission standards for nitrogen oxide, a pollutant implicated in the long-range transport, transboundary air pollution problem. The principles do provide, however, for accelerating acid rain research. [11]

B. Canadian Perceptions and Policies

Acid precipitation is of great concern to Canadian citizens [12] and officials of the Canadian Government. Environment Minister John Roberts characterized the phenomenon as "the most serious enviromental threat ever to face the North American continent." [13] Ray Robinson, former Director of Canada's Environmental Protection Service, described the seriousness of the problem and the perceived threat to Canadian economic interests as follows:

> "A significant portion of Canada's economic activity and a wide range of societal interests are threatened by acid rain. The government of Canada would like to be able to reassure those interests, to be able to demonstrate that the phenomenon is not serious. But the weight of emerging scientific evidence not only makes such reassurance impossible, it drives home the urgency of finding solutions before a great deal more irreversible damage is done." [14]

Since the signing of the 1980 Memorandum of Intent, Canada has taken a number of steps to demonstrate the seriousness of its concern over acid deposition. Recently, the Canadian government pressed the United States to reduce SO_2 emissions in the course of negotiations by proposing a draft agreement which required parallel reductions in both countries. [15] The Province of Ontario has petitioned the Administrator of EPA opposing relaxations of SO_2 limits in State Implementation Plans and urging enforcement of existing limits. [16]

In addition, Canada has taken steps to control its own sources which contribute to transboundary pollution, and to improve Canadian legal mechanisms for control. A recent release from the Canadian Embassy notes:

> Canada has . . . begun to deal with its principal sources of transboundary air pollution. The large smelter at Sudbury, Ontario has been served with a control order to reduce emissions over a specified time, and a study has been launched to determine the greatest additional reductions that would be feasible. Ontario Hydro, the other major source of acid precursor pollutants in Central Canada, is beginning a series of steps aimed at reducing emissions 43% by 1990. [17]

Canadians perceive acidic precipitation to be a proven problem -- one which is jointly shared by Canada and the U.S., and which requires immediate bilateral action for its solution. The report to Parliament of the Canadian House of Commons Subcommittee on Acid Rain reflects Canadian dissatisfaction with the direction of current U.S. research and control efforts:

> The Memorandum of Intent signed between Canada and the United States is a commitment from both countries not only to enter into formal treaty negotiations but to also combat acid rain immediately. The United States should be reminded periodically that we in Canada feel bound by the Memorandum and we expect a reciprocal commitment from the United States, notwithstanding a change of Administration. To put any other interpretation on this document would jeopardize future negotiations between our two countries. We have treated our obligations seriously under this document . . . To date it appears that the United States is not approaching the problem in the same spirit. The Sub-committee urges the United States to improve its legislation, regulations and enforcement now so that acid rain-causing emissions will be substantially reduced, and that it not await the signature of a transboundary air pollution agreement. 18/

Canadian perceptions of the gravity of the problem were highlighted when Prime Minister Trudeau raised the issue with a newly-elected President Reagan on his first visit to Canada. 19/

II. Diplomatic/Legal Context of the U.S.-Canadian Transboundary Air Pollution Issue.

A. Toward a Transboundary Agreement: History of U.S.-Canadian Policies Leading to the Memorandum of Intent

While no agreement between the United States and Canada directly governs the control of transboundary pollution, the two countries have a long history of cooperation on environmental affairs and joint commitments to principles for controlling transboundary pollution. Moreover, the United States has been recognized as a world leader in the international environmental field, in large part as a result of actions which bear on transboundary pollution issues.[20].

Cooperation on transboundary pollution issues between the United States and Canada began as early as 1909, when the two countries concluded the Boundary Waters Treaty. The treaty provided in part:

> ". . . boundary waters and waters flowing across the boundary shall not be polluted on either side to the injury of health or property."[21]

Under the treaty, the countries established the International Joint Commission (IJC). The IJC is an impartial body charged with monitoring progress under the agreement, and can, within its mandate, consider any question submitted to it by the two governments, including those dealing with air pollution.[22]

In the 1920's and 1930's, a dispute over transboundary air pollution from a smelter in Trail, British Columbia was brought to arbitration, and damages were awarded.

The United States as well as Canada has expressed commitments to the principles set out by the Trail Smelter case in more recent years. The 1972 Declaration of the U.N. Conference on the Human Environment, supported by both countries, incorporates the earlier "Trail Smelter" doctrine. Principle 21 of the Declaration states:

> "States have, in accordance with the Charter of the United Nations and the principles of international law, the sovereign right to exploit their own resources pursuant to their own environmental policies,and the *responsibility to insure that activities within their jurisdiction or control do not cause damage to the environment of other states or areas beyond the limits of national jurisdiction*." (Stockholm Declaration on the Human Environment, U.N. Doc. A/Conf. 48/14). (Emphasis added.)

During the 1970's, the U.S. and Canada also conducted major negotiations over water pollution to the Great Lakes, culminating in the 1978 Great Lakes Water Quality Agreement. The accord not only authorizes specific steps to achieve agreed-upon water goals, but also includes an agreement to consult on appropriate remedial programs in cases where air pollution causes significant contributions to Great Lakes pollution. [23]

In 1978, the United States and Canada also began to directly address the issue of transboundary air pollution caused by long-range transport by establishing "Bilateral Research Consultation Group (RCG) on the Long Range Transport of Air Pollutants (LRTAP)." The RCG has published two reports to date which characterize acid precipitation as a serious international problem and the call for further research. [24] In the same year the U.S. Congress also passed a joint resolution exhorting the President to make efforts to negotiate a cooperative agreement with Canada to address transboundary air pollution.

During the year 1978, the Congress also passed a joint resolution exhorting the President to make every effort to negotiate a cooperate agreement with Canada to address transboundary air pollution. The resolution states, in part:

> "(b) It is the sense of the Congress that the President should make every effort to negotiate a cooperative agreement with the Government of Canada aimed at preserving the mutual airshed of the United States and Canada so as to protect and enhance air resources and insure the attainment and maintenance of air quality protective of public health and welfare.
>
> (c) It is further the sense of the Congress that the President, through the Secretary of State working in concert with interested Federal agencies and the affected States, should take whatever diplomatic actions appear necessary to reduce or eliminate any undesirable impact upon the United States and Canada resulting from air pollution from any source." [25]

After the resolution of 1978, Congress and the Executive Branch proceeded in tandem, taking actions which indicated that both branches of the United States Government recognized transboundary air pollution--in particular, acid precipitation--as a serious problem that required bilateral remedial actions.

In 1979 the United States became a party to the Economic Commission for Europe "Convention on Long-Range Transboundary Air Pollution" and reaffirmed its commitment to controlling transboundary air pollution. Article 2 states that the "contracting parties are determined to protect man and his environment against air pollution and shall endeavor to limit and, as far as possible, gradually reduce and prevent air pollution including long-range transboundary air pollution." 26/ Canada was a party to this convention as well.

During the year 1979, informal bilateral discussions with Canada also began. They concluded with a Joint Statement on Transboundary Air Quality which recognized the ECE convention and expressed the countries' intent to move beyond the informal stage to develop a bilateral agreement on transboundary air pollution. 27/

B. The Memorandum of Intent

Shortly following the Joint Statement in 1980, the United States concluded the U.S.-Canada Memorandum of Intent concerning Transboundary Air Pollution (MOI). The MOI states, in part, that the Government of Canada and the Government of the United States of America intend:

1. to develop a bilateral agreement which will reflect and further the development of effective domestic control programs and other measures to combat transboundary air pollution;

2. to facilitate the conclusion of such an agreement as soon as possible; and,

3. pending conclusion of such an agreement, to take interim actions available under current authority to combat transboundary air pollution. [28]

The MOI specifically outlined the interim actions including control measures to which the two governments agreed. The MOI states:

> To combat transboundary air pollution both Governments shall:
>
> (a) develop domestic air pollution control policies and strategies, and, as necessary and appropriate, seek legislative or other support to give effect to them;
>
> (b) promote vigorous enforcement of existing laws and regulations as they require limitation of emissions from new, substantially modified and existing facilities in a way which is responsive to the problems of transboundary air pollution; and
>
> (c) share information and consult on actions being taken pursuant to (a) and (b) above. (Emphasis added.) [29]

The MOI also established five Work Groups to assist in preparing and conducting negotiations. The Work Groups were given a comprehesive mandate to assess the existing and potential impacts of LRTAP and acid rain, the possible control strategies available to the two governments, and the costs of these strategies. The Work Groups published interim reports in 1981 and are, at this writing, working on the final reports originally due in January, 1982.

C. The MOI: Status of Commitments to Interim Actions

The precise legal status of the MOI and, in particular, the commitments to interim actions contained therein is unclear. Within the United States and Canada international law is considered a part of domestic law. [30] The MOI clearly expresses "common determination to combat transboundary air pollution in keeping with existing international rights, obligations, commitments and cooperative practices including those set forth in the [1909 Boundary Waters Treaty, the 1972 Stockholm Declaration, the 1978 Great Lakes Agreement and the 1979 ECE Convention.]" 31/ However, as a practical matter it would be difficult to use international law referred to in the MOI as a means to compel interim control measures. (See Section IV.C. for a discussion of remedies under international law.) The MOI itself is not a binding international agreement; as a joint statement of government intention, it draws some force from the general good will between the U.S. and Canada and the related expectation of good-faith implementation.

The MOI commitment to take all possible actions to reduce transboundary pollution through the domestic pollution control system is sweeping in nature. However, its terms have been subject to varying interpretations. The Province of Ontario, for example, has initiated new control programs, attributing their necessity, in part, to the mandate in the MOI. The Province has asserted that the MOI requires the U.S. to take similar steps, or at a minimum, not allow emissions to increase. 32/ But the U.S. EPA has not interpreted commitment to take interim actions available under "current authority" to require control efforts through Clean Air Act provisions pertaining to interstate and international air pollution. 33/ A 1981 EPA administrative proceeding issued a ruling that the international impacts need not be considered in decisions over delaying SIP emissions limitations. The agency concluded that the MOI mandate to "promote vigorous enforcement of existing laws and regulations" is satisfied so long as emission limitations are adequate to assure compliance with ambient air quality standards locally, even if emissions increase from current levels. 34/ Thus the United States and Canada currently adopt very different views of what "good faith" requires under the

MOI. However, as a practical matter, Canada can do little to compel the U.S. to take actions consistent with its interpretation of the MOI. [35]

III. Mechanisms to Control Transboundary Pollution under the Clean Air Act

Like the interstate provisions of the Clean Air Act, the transboundary control provisions of Section 115 rely on the SIP process as a means to achieve control. However, unlike the interstate provisions Section 115 does not restrict itself to the control of criteria air pollutants for which there are NAAQS. Section 115 allows control of any air pollutant. In addition, the section may be interpreted to allow for the control of transboundary pollution caused by acidic deposition. The utility of Section 115 for the control of pollution caused by LRTAP in the U.S.-Canadian context is in large part determined by whether and how the Administrator of EPA chooses to implement the Section. However, Section 115's reliance on the SIP process could make implementation of control measures for non-criteria pollutants more difficult.

A. Section 115

The Congress adopted Section 115 of the Clean Air Act to provide a means to deal with transboundary pollution problems. [36] The section is brought into play by the Administrator of EPA on his own initiative or at the request of the Secretary of State. The Secretary must allege, or the Administrator must determine, that "he has reason to believe" that pollution emitted in the United States causes or contributes to air pollution which "may reasonably be anticipated to endanger public health or welfare in a foreign country". The section provides that the Administrator can make such a determination "upon receipt of reports, surveys or studies from any duly constituted international agency"; however, it contains no such language as to how the Secretary of State will be apprised of a problem.

Once the Secretary of State has requested the Administrator to activate the 115 process or the Administrator has made a finding of "endangerment", the statute requires the Administrator to give formal notice to the Governor(s) of the state(s) in which the emissions causing the pollution originate. The Administrator must take this step, however, only if he "determines the foreign country receiving the pollution has given the United States essentially the same rights with respect to the prevention or control of air pollution occurring in that country as is given that country by [section 115]." Thus the Administrator must make a determination of reciprocity to trigger the mandatory duty under Section 115.

Section 115 relies on the SIP revision process as the means to abate international pollution prohibited by the Section. Section 115 provides that notice given to the Governor constitutes a finding with respect to Section 110(a)(2)(H) -- i.e., that the state implementation plan is inadequate to comply with the requirements of the Clean Air Act and must be revised.

B. Utility of Section 115 for the Control of LRTAP

1. Control of Non-Criteria Pollutants and Acidic Deposition

Section 115(a) provides that the Administrator may activate the SIP process when he "has reason to believe that any air pollutant or pollutants emitted in the United States causes or contributes to air pollution which may reasonably be anticipated to endanger public health or welfare in a foreign country" (emphasis added). Welfare effects are defined by the Act to include effects on soils, water, crops, vegetation, manmade materials, wildlife, weather and visibility (Section 302(h)). Therefore, the section could be construed to allow for the control of transboundary pollution caused by acidic deposition.

2. Implementation of the Section

The language of Section 115 falls short of giving the Administrator clear guidelines on how to implement the section. It does not detail the procedures for the Administrator to follow in identifying which state or states are the source of the transboundary emissions, and provides no guidance as to how to allocate control responsibilities when emissions from more than one state create transboundary problems.[37] The legislative history of Section 115 is also silent on these issues.

In practice, voluntary activation of Section 115 under the present Administration is unlikely. The Agency has historically been reluctant to face the allocational dilemma in the interstate controversies, it might also be reluctant to address such a politically volatile issue in the context of protecting the health and welfare of another country.

Moreover, the EPA has contended that there are no models which can be used for regulatory purposes to assess the long-range transport impacts of SO_2 emissions on ambient concentrations of sulfates or acidic deposition. Action under 115 could also be perceived as a limiting of the U.S. bargaining position at the table with Canada.

C. Activity under Section 115

Under the previous Administration, actions were taken by then-EPA Administrator Douglas Costle to activate Section 115 to control acidic deposition in Canada. However, it appears that the process set in motion by Administrator Costle has stopped. The current Administrator has not notified any state pursuant to Section 115. There have been no court decisions indicating whether the actions of Costle are legally binding on the present Administrator and require her to take control actions through the SIP process.

1. The Mitchell Request

The Canadian Clean Air Act was amended following unaminous vote in both Houses of Parliament in December 1980 to provide clear authority for the Canadian Federal Government to take steps to control the emission of pollutants affecting another country. The amendment was designed primarily to allow "mutual recourse between Canada and the United States". [38/]

On December 23, 1980, Senator George Mitchell of Maine sent letters to the Administrator of EPA and the Secretary of State, [39/] calling their attention to legislation recently enacted by the Canadian Parliament and reports prepared by the International Joint Commission, and the United States - Canada Research Consultation Group on Long-Range Transboundary Air Pollution (RCG). The Senator, reviewing the information in the reports, concluded with a strongly-worded request that action be taken under section 115:

> "In my judgement, these reports provide ample basis for a determination under Section 115(a) that air pollutants emitted in the United States are causing or contributing to air pollution that may reasonably be anticipated to endanger health or welfare in Canada. Since Canada also enacted reciprocal legislation, I believe you are now required to

> notify appropriate States to revise their implementation plans [SIP] as necessary to prevent or eliminate the endangerment caused by acid precipitation." 40/

2. The Costle Response: Determination to Activate Section 115

On December 24, 1980, the Department of State announced that the United States would evaluate the Canadian legislation to determine whether it provides essentially the same rights as Section 115 of the U.S. Clean Air Act. 41/

On January 13, 1981, Administrator Costle of EPA responded to the Mitchell request and sent letters to both the Senator and the Secretary of State announcing his findings. He released his findings to the media in an EPA press release on January 16, three days before his departure as Administrator of EPA. 42/

In his response to Senator Mitchell, relying on the information presented by the IJC report, Costle states that:

> "EPA has concluded that acid deposition, often referred to as acid rain, presents a genuine threat to our environmental well-being both in the U.S. and Canada." 43/

Outlining the seriousness of the problem, he states among other things that acid deposition "can and has destroyed lake and stream ecosystems in Canada and the U.S." Costle then specifically stated that Section 115 was broadly drafted to address these issues:

> "These kind of impacts are within the range of impacts covered by Section 115....[The] Section is broadly drafted to encompass all forms of air pollution-related endangerment to public health or welfare and it is not limited to interference with U.S. air quality standards or significant deterioriation programs as is Section 126 of the Clean Air Act. 44/

Thus, Costle determined that the SIP revision process could be used to reduce emissions under Section 115 to prevent environmental damage caused by the deposition of air pollutants. He then investigated whether the preconditions for activation of Section 115 had been met. First, he determined that:

1) The International Joint Commission is a duly constituted agency under Section 115.

2) The October 1980 Seventh Annual Report on Great Lakes Water Quality of the International Joint Commission shows that the IJC has found acid deposition to result in significant harm in both the U.S. and Canada and that emissions sources in both countries contribute to this problem through the long range transport of air pollution. Thus emission sources in the US caused or contributed to a serious enough problems in Canada to warrant Section 115 action. [45]

For Costle to bring Section 115 into play, however, a further determination was needed on whether reciprocity existed between the U.S. and Canada as required by Section 115(c). Costle found reciprocity but qualified his findings.

Costle concluded that a finding of reciprocity would require 1) determining whether there was legislative authority to grant the United States essentially the same rights, and 2) determining that as applied the Canadian law grants reciprocal rights. Costle specifically drew attention to the differences in procedures between the U.S. and Canadian systems, but found that these procedural differences were not a barrier to a finding of reciprocity. [46] Costle's determination is best laid out in his letter to then Secretary of State, Edmund Muskie:

> As required by the Clean Air Act, I have completed my review of the Canadian legislation. After consultation with the Department of State, I have concluded that the Canadian

> legislation provides the Government of Canada with authority to give the United States essentially the same rights as Section 115 of the Clean Air Act gives to Canada. As with most legislation it is possible that the Canadian legislation could in the future be interpreted or implemented in a way that the United States would conclude that it was not being given essentially the same rights as are provided under Section 115. Thus, it not possible to make a permanently binding determination... [One must look to the legislative authority and how it is applied.] This second aspect of EPA's determination is necessarily a dynamic one which will continue to be influenced by Canadian action now and in the future.[47]

Thus, having made the requisite determinations to activate Section 115, the Administrator instructed his staff to begin work to identify which states should receive formal notification and to lay the groundwork to assist these states in appropriately revising their SIPs.[48] David Hawkins, then Assistant EPA Administrator for Air, Noise and Radiation, sent a memorandum to the EPA's Office of Air Quality Planning and Standards (OAQPS). It requested the staff to "proceed to develop information and recommendations for next Administrator as to which states might be appropriately notified [under Section 115][49] and a suggested timetable for such notifications."

There have been no indications that the work initiated by Costle is proceeding under the new EPA Administrator, Anne Gorsuch. As yet, Section 115 notification has not been given to any State and EPA appears not to have continued the process leading up to notification.[50] EPA has issued no statement, however, which reverses the Costle determinations. No court has yet reviewed the legal significance of Costle's action.[51]

IV. Remedies for Transboundary Pollution

A. Recourse under the Clean Air Act

The initial decision to activate section 115 is clearly discretionary. Thus, the statute does not explicitly provide a cause of action which Canada or others could pursue in domestic courts to remedy transboundary pollution. An argument can be made, however, that the actions of former Administrator Costle "activated" Section 115 and created a legal obligation for the present Administrator to revise SIPs to control acidic deposition in Canada. Section 115 prescribes that the Administrator shall formally notify the state(s) whenever he makes a determination of endangerment pursuant to 115(a) and reciprocity pursuant to his duty under §115(c). Thus, it could be claimed that Costle had triggered a mandatory duty to give formal notification to the appropriate states by the following actions:

1) His letter to Secretary of State Muskie in which he transmitted his determination that U.S. sources were causing pollution prohibited by Section 115(a) and that the section should be activated.

2) His response to Senator Mitchell's request in which he transmits the same findings.

3) The press release in which he announced his findings to the public with regard to Section 115.

Section 304 of the Clean Air Act authorizes "any person" to bring suit to compel the Administrator to perform non-diseretionary duties under the Act. [52/] Section 302(e) provides that:

> The term "person" includes an individual, corporation partnership, association, State, municipality, political subdivision of a State and any agency, department, or instrumentality of the United States and any officer, agent or employee thereof. 42 USC§ 7602(e).

It is clear that under the citizen suit provision a number of different groups could bring suit to compel action under Section 115 -- among them are environmental organizations and northeastern States who might "benefit" from activation of the Section. The eligibility of Canada or the Province of Ontairo to bring suit is less clear. Section 302(e) does not specifically mention foreign governments and no court has addressed the question of whether foreign governments are considered to be "persons" under Section 302(e) and 304. 53/ However, standing would not be a barrier to litigation by plaintiffs having similar interests as Ontario or Canada.

Should a suit be initiated under Section 304 to compel Administrator Gorsuch to notify States under Section 115 to revise their SIPs, a court could be faced with determining whether Costle's findings triggered such a duty. The courts could, in addition, be faced with the question of whether these findings were arbitrary and capricious or in excess of statute, and as such, unlawful.

The present Administrator would have the burden of proving to a court that the previous Administrator had acted unlawfully in reaching his determinations concerning endangerment and reciprocity. This would be a difficult burden of proof to overcome since the statute gives the Administrator great discretion, and courts as a rule pay great deference to administrative determinations.

It would be difficult for the present Administrator to establish that Costle's endangerment determination was arbitrary and capricious for two reasons. First, Section 115 gives the Administrator the authority to make a determination that a pollution problem exists which may be controlled pursuant to the provision whenever he has "reason to believe" that pollution "may be reasonably anticipated to endanger public health or welfare in a foreign country." The Section does not require that harm be conclusively established to activate the transboundary provision. Second, the Administrator had a factual basis for his determinations -- the International Joint Commission.

The Administrator would also have a difficulty in convincing a court that Costle's reciprocity determination was unlawful. Congress has left much to the discretion of the Administrator as to what standards should be applied to determine whether reciprocity exists. The present Administration appears to have adopted a position that exact equivalency between Canadian and United States statutes is not required to find reciprocity; rather, a "functional equivalency" is sufficient for reciprocity. Secretary of the Interior James Watt recently determined that there was reciprocity between Canada and the U.S. as required under the Mineral Lands Leasing Act of 1970 (MLLA) (30 U.S.C. § 181). [54/] Like the Clean Air Act, MLLA did not outline the elements of reciprocity. The Secretary found reciprocity even though Canada limits foreign ownership -- and hence U.S. ownership -- of oil and gas production leases to 50 percent. No such percentage limitation exists with respect to Canadian ownership under the MLLA.

Section 21.1 of the Canadian Clean Air Act provides a mechanism for the establishing new control requirements where emissions in Canada "create or contribute to" air pollution "that may reasonably be expected to constitute a significant danger to the health, safety or welfare of persons" in another country. The key phrase "welfare of persons" is undefined. The definition of "air pollution" elsewhere in the Canadian Clean Air Act suggests that damages to animal and plant life are considered distinct from the welfare of persons. Canadian Clean Air Act § 21.1(1)(b). 55/ Nevertheless, the Canadian regulatory structure is far less rigorous than that of the U.S. If they chose, the Environment Minister and Governor-in-Council (the Cabinet) would have ample latitude to give effect to the legislative intent and interpret "welfare" to include environmental effects and provide what can be termed "functional" reciprocity. The legislative history of the amendments to the Clean Air Act, as well as recent control actions taken by the Province of Ontario, indicate that Canada is willing to so interpret the legislation. 56/

B. Recourse Through Diplomatic Channels

The Canadian Government could choose to link the transboundary issue to other areas of bilateral concern in an effort to encourage adoption of policies to reduce the long-range transport of air pollution and acidic deposition. Canada is an important ally and neighbor. Our involvement with Canada is probably greater than with any foreign country, and the involvement is unique. Deputy Assistant Secretary of State for European Affairs, Raymond Ewing, described some of the important links between the two countries as follows:

> "The two-way trade between the two countries currently is some 77 billion, twice as much as American trade with Japan. The trade between the two countries is greater than that between the United States and all of the countries of the European Community. We are allies together in NATO. We are members together in the Organization for Economic Cooperation and Development. Bilaterally, we have a unique military joint command, NORAD, the North American Aerospace Command." 57/

C. Recourse in the Domestic Courts under Common Law

Commentators and international organizations are devoting increasing attention to the potential role of suit in domestic courts as an avenue for redressing transboundary air pollution problems. 58/ However, there are practical difficulties using judicial action to redress international problems. Adjudication by a domestic court on the merits of transboundary pollution concerns can imply an infringement on the foreign sovereignty. 59/ As a rule, nations prefer to deal with significant international problems through the negotiations process, where any resolution will by definition be mutually agreeable.

1. Common Law Actions Under State Law

Use of common law in transboundary pollution disputes has to date been limited largely to actions in nuisance. 60/ Two cases involving short-range transboundary pollution, Michie v. Great Lakes Steel 61/ and Ohio v. Wyandotte Chemical Corporation 62/ are recent examples. These cases evidence the capacity of domestic courts to resolve transboundary environmental disputes where the harm occurs relatively near the pollution source.

Establishing causation is, of course, likely to present a more important barrier in lawsuits concerning effects associated with long-range air pollution. In particular, it would be difficult to document effects and relate them to emissions from identifiable sources. The Michie precedent, permitting collections of sources to be found jointly and severally liable for pollution damages, could in theory enhance the feasibility of such a suit, at least under Michigan law. Following this precedent, plaintiffs could proceed against a collection of sources contributing, for example, to pollution loading in an area where emissions have been demonstrated to affect acid-sensitive regions. However, beyond the legal obstacles, the costs of monitoring, air quality modeling, scientific advice and expert testimony could present major problems in any suit to redress for long-range air pollution problems.

2. The Federal Common Law of Nuisance

The doctrine of nuisance under Federal common law appeared until recently to afford a promising cause of action for private parties attempting to obtain compensation for damage due to transboundary pollution from the U.S. [63] However, two recent Supreme Court decisions, Milwaukee v. Illinois [64] and Middlesex County Sewerage Authority v. National Sea Clammers Association [65] have greatly diminished the applicability of the doctrine. In Milwaukee the Court ruled that the Federal Water Pollution Control Act preempts any cause of action based on the federal common law of nuisance where the defendant's discharges are subject to standards promulgated under the Act. The Court's rationale in City of Milwaukee was that Congress, in establishing a "comprehensive" regime for regulating water pollution, had occupied the field legislatively and displaced the authority of the federal courts to attach additional legal requirements onto existing administrative rules. The Court went even further in Middlesex County, ruling that the FWPCA preempts all claims under the federal common law of nuisance seeking redress for any form of water pollution governed by the Act or, to the extent not governed by the FWPCA, governed by the Marine Protection, Research, and Sanctuaries Act of 1972, 33 USC § 140 a *et seq*. Thus, the citizen suit provisions of the Clean Water Act now provide the sole remedy available to private citizens seeking relief in court for damages caused by water pollution governed by the FWPCA.

The implications for future common law nuisance actions to control transboundary air pollution are somewhat unclear. The viability of an action brought under federal common law will in large part depend on whether courts characterize the Clean Air Act as "comprehensive", as did the court for the FWPCA. No court, as yet, has directly addressed the issue. 66/

Apart from these substantive considerations, the procedural complications in a transboundary nuisance action can also be severe, especially for the plaintiff who fails to select the proper forum. 67/

D. Recourse under International Law

The principles of international environmental law offer the central guidance concerning the responsibility of one nation to control pollution causing damage to another. Several international organizations have attempted to formulate general principles concerning the responsibility of states for extraterritorial damages caused by pollution. These principles are, in general, expansions of the ancient Roman maxim sic utere tuo ut alienum non laedar (use your property so as not to injure that of another). Probably the most influential international statement on this subject is the 1972 Declaration of the United Nations Conference on the Human Environment in Stockholm. Analogous principles may be found in the more developed area of international water pollution. 68/ However, it is unlikely that accepted notions of international law can alone compel major modifications in the energy and pollution control policies of industrialized nations. There are four major reasons.

First, the principles are not sufficiently well defined to support ready application to specific controversies. As commentators have noted, general statements concerning the duty of states to avoid actions adversely affecting the environment of other nations are not responsive to the difficult questions concerning precisely what types of conduct are unacceptable. 69/ Present international legal doctrines do not offer sufficient guidance to permit nations to define with specificity the point at which an interest in industrial development is outweighed by concerns surrounding the environmental effects of transfrontier pollution.

Second, even if a given controversy is sufficiently clear-cut to permit their application, no effective mechanism presently exists for enforcing international legal doctrines. No international agency is empowered to give force to international environmental principles not incorporated into binding agreements. 70/

Third, in the few existing applications of international environmental law where the nations involved have consented to be bound by the decision of a neutral tribunal, claimants have been required to demonstrate that specific identifiable sources have caused environmental injury. The incomplete scientific understanding of the intricacies of the long-range transport phenomenon make it unlikely to be able to assign responsibility for distant transboundary environmental effects under such a standard.

Finally, the present international legal framework does not effectively foster preventive action. General principles concerning the responsibility of nations to provide compensation for the damages caused by transboundary pollution are useful in allocating expenses, and may have some deterrent value, but they are of only limited utility in avoiding permanent environmental damage.

FOOTNOTES

1/ This chapter is the compilation and synthesis of 1) staff research and 2) information contained in a report prepared under contract for OTA by Gregory Wetstone. The contractor report is entitled "Long Range Air Pollution across National Boundaries: Resources in Law and Policy" (January, 1982).

2/ Canadians have cited evidence that nearly 80% of wet acidic deposition in some sensitive areas of Canada is generated by emissions from sources in the United States. Province of Ontario "Submission to the United States Environmental Protection Agency Opposing Relaxation of SO_2 Emission Limits in States Implementation Plans and Urging Enforcement" March 12, 1981.

3/ See for example, the petition of the Province of Ontario before the EPA to review relaxation of emission limits of a number of sources. Ontario argues that such relaxations increase acidic deposition in Canada and increase damage associated with the phenomenon. Such a policy, Ontario argues, the spirit if not the letter of the U.S. and international law to control transboundary air pollution. Id.

4/ "Canada May End Talks with U.S. for Accord to Combat Acid Rain," The Wall Street Journal, June 16, 1982, pp. 1, 7.

5/ "Bennett, EPA Air Program Nominee, Calls Controls for Acid Rain Premature," Env. Rep. (BNA), July 24, 1981, p. 411; and Cohn, R., "EPA Chief Says Clean Air Laws Have Hurt Economy," Philadelphia Inquirer, June 23, 1981 at 5-A. Consistent with this view, the EPA has requested accelerated acid rain resarch. For an updated discussion, see "Reagan Administration Drafts, Redrafts Specific Clean Air Act Amendments,' Air and Water Poll. Rep. (August 31, 1981).

6/ National Journal, p. 456 (March 12, 1982.)

7/ President Carter's Environmental Message to Congress, August 2, 1979, reprinted in Environment Reporter, (August 24, 1979).

8/ 42 USC §8901 et seq.

9/ Section 115 of the Clean Air Act. For a discussion of EPA activity under Section 115, see pp. 14-17 infra.

10/ White House, Office of the Press Secretary, Background Briefing "Following Visit of Pierre Elliot Trudeau, Prime Minister of Canada," July 10, 1981.

11/ Press release from Environmental Protection Agency Administrator Ann Gorsuch, August 5, 1981. For an environmental critique of the August 5 principles, see National Clean Air Coalition, Breaking the Code: Analysis of the Administration's "Principles" for Amending the Clean Air Act, August 27, 1981.

These principles are not necessarily the final statement of the Administration position on reauthorization of the Clean Air Act. The door was left open for the sumbission of a more complete legislative package if the Administration deems such a move necessary. Also, it is unclear how these general principles will be interpreted in Administration lobbying efforts.

12/ A recent poll conducted by the Province of Ontario indicated that 70% of the population is aware of acid rain and believes it to be a serious problem. "Acid Rain: Who Will Save Our Lakes?" McLean's, June 20, 1980.

13/ Press Release from Environment Canada, "Speech by Honorable John Roberts before the International Association of Water Pollution Research," June 25, 1980.

14/ Address by Ray Robinson, Assistant Deputy Minister for Environmental Protection, to Conference on Acid Rain, State University of New York at Buffalo, May 2, 1981.

15/ Canada submitted a draft Transboundary Air Pollution Agreement to U.S. negotiations during the February 24, 1982 negotiation session. The draft called for parallel reductions in SO_2 of 50 percent by the U.S. and Canada. The Canadian Environment Minister announced that the proposed reductions reflect the findings of U.S.-Canadian research groups established by the Memorandum of Intent that pollutant leadings should be reduced to 20 kg/ha/yr of sulphate in order to protect moderately sensitive lakes and streams. Press Release, Canadian Embassy, Public Affairs Division, Highlights of Canadian Negotiating Position on Acid Rain, February 24, 1982.

16/ Province of Ontario Submission to the United States Environmental Protection Agency Opposing Relaxation of SO_2 Emission Limits in States Implementation Plans and Urging Enforcement, March 12, 1981.

17/ Canadian Embassy Release, Long Range Air Pollution and Canada-U.S. Relations, Washington, D.C. (February 1982).

18/ House of Commons Subcommittee on Acid Rain of the Standing Committee on Fisheries and Forestry, Still Waters, p. 92 (Ottawa, 1981).

19/ White House, Office of the Press Secretary Background Briefing, "Following Visit of Pierre Elliot Trudeau, Prime Minister of Canada," July 10, 1981.

20/ Prepared Statement of James L. Buckley, Under Secretary of State for Security Assistance, Science and Technology before the House Subcommittee on Human Rights and International Organizations of the Committee on Foreign Affairs, March 30, 1982.

21/ ART IV, Boundary Waters Treaty of 1909, U.S.T. 548, reprinted in Int'l Envir. Rep. 31:0401.

22/ Stein, R., and Flemming, B., The Use of Section 115 of the Clean Air Act to Control Long Range Transport of Air Pollution between the United States and Canada, a Report for Environmental Canada and the United States Environmental Protection Agency (Environmental Mediation International, Inc.), Washington, D. C. (1981), p. 28. The IJC has dealt with transboundary air pollution issues in the Detroit/Windsor area. In its 1972 report it is stated that air quality objectives had been developed to protect the public health and welfare of both sides of the border.

23/ Article VI, 1978 Agreement Between the United States and Canada on Great Lakes Water Quality, reprinted in Int'l Envir. Rep. 31:0604.

24/ United States-Canada Resarch Consultation Group on the Long-Range Transport of Air Pollutants, "The LRTAP Problem in North America: a preliminary overview," pp. 24-25, 1979.

25 42 U.S.C. § 7415 note, P.L. 95-426, Title VI, § 612, October 7, 1978.

26/ United Nations Economic Commission for Europe, Convention on Long-Range Transboundary Air Pollution, Geneva, November 13, 1979.

27/ Joint Statement on Transboundary Air Quality by the Government of Canada and the Government of the United States of America concerning Transboundary Air Pollution, August 5, 1979.

28/ Memorandum of Intent between the Government of Canada and the Government of the United States of America concerning Transboundary Air Pollution, August 5, 1980.

29/ Id.

30/ Stein, "The Legal Considerations," p. 66, contained in Clean Air for North Americans: Acid Rain in Canadian American Relations, Canada Studies Program, Columbia University, 1981.

31/ Memorandum of Intent between the Government of Canada and the Government of the United States of America concerning Transboundary Air Pollution, August 5, 1980.

32/ Address by Keith Norton, Ontario Minister of the Environment to the Conference on Acid Rain, State University of New York at Buffalo, May 1, 1981.

33/ Mosher, L., "Congress May Have to Resolve Stalled U.S.-Canadian Acid Rain Negotiations," National Journal, p. 456, March 13, 1982; "Canada May End Talks With U.S. for Accord to Combat Acid Rain," The Wall Street Journal, June 16, 1982, pages 1-7. column 3.

A number of recent or proposed U.S. actions are expected to increase rather than decrease transboundary air pollution. Examples include administrative relaxations of emission limitations for major mid-western sulfur dioxide emitters, and the Administration's principles for revisions to the Clean Air Act, which include a number of relaxations in relevant pollution control requirements.

34/ 46 FR 37645 (July 22, 1981).

35/ "Canada May End Talks with U.S. for Accord to Combat Acid Rain", The Wall Street Journal, Section 1, p.7, Column 3.

36 The Section states, in part:

> (a) Whenever the Administrator, upon receipt of reports, surveys or studies from any duly constituted international agency has reason to believe that any air pollutant or pollutants emitted in the United States cause or contribute to air pollution which may reasonably be anticipated to endanger public health or welfare in a foreign country or whenever the Secretary of State requests him to do so with respect to such pollution which the Secretary of State alleges is of such a nature, the Administrator shall give formal notification thereof to the governor of the state in which such emissions originate.
>
> (b) The notice of the Administrator shall be deemed to be a finding under section 110(a)(2)(H)(ii) which requires a plan revision with respect to so much of the applicable implementation plan as is inadequate to or eliminate the endangerment referred to in subsection (a). Any foreign country so affected by such emission of pollutant or pollutants shall be invited to appear at any public hearing associated with any revision of the appropriate portion of the applicable implementation plan.
>
> (c) This section shall apply only to a foreign country which the Administrator determines has given the United States essentially the same rights with respect to the prevention or control of air pollution occurring in that country as is given that country by this section.

37/ In the case of long-range transport since emissions from sources in multistate regions can contribute to a transboundary problem, this will involve the use of air quality models and other analytic tools. Such models are not capable of identifying the effects of single emission sources at great distances. The models can be used, however, to relate emission source areas to areas of environmental impact or air quality degradation.

As was noted previously, states, in establising emission limitations under SIPs, do not take into account the multiple source impacts far downwind; rather, emission limitations are set to ensure compliance with NAAQS within the locale of the source. This focus of the SIP process would make administration of any program to control LRTAP's and acidic deposition cumbersome in the absence of a change in EPA policies with regard to a state's duties under the interstate provisions to consider impacts caused by long-range transport.

38/ House of Commons Subcommittee on Acid Rain of the Standing Committee on Fisheries and Forestry, *Still Waters* p. 92 (Ottawa 1981). As was noted in Environmental Canada's Annual Report, the Canadians believe the amendment "provides reciprocity required by Section 115 of the U. S. Clean Air Act which enables U.S. federal authorities to initiate a process to require state governments to reduce emissions adversely affecting Canada." Environment Canada *The Clean Air Act Annual Report, 1980-1981*, p. 1 (October 1981).

39 Letter from Senator George Mitchell to EPA Administrator Douglas Costle, December 23, 1980 and Letter from Senator George Mitchell to Secretary of State Edmund Muskie, December 23, 1980.

40/ Mitchell letter to Costle.

41/ Letter from Douglas Costle, EPA Administrator, to Edmund Muskie, Secretary of State, January 13, 1981.

42/ Press Release, U. S. Environmental Protection Agency Environmental News, January 16, 1981.

43/ Letter from Douglas Costle, EPA Administrator, to Senator George Mitchell, January 15, 1981.

44/ *Id.*

45/ *Id.*

46/ Costle noted:

> "The principal difference in the two statutes is the detailed procedural and substantive requirements applicable to the State plan revision process under the U.S. Clean Air Act as opposed to the more general requirement in the Canadian legislation for provincial consultation and reasonable efforts to secure action by the provincial government. In my judgment, that difference does not significantly restrict the ability of the Government of Canada to provide essentially the same rights to the United States. The Canadian requirement for federal consultation and efforts to procure provincial action fills the same role as the State plan revision process in the U.S. system. *Consequently, I have concluded that, despite the differing process at the State and provincial levels, the Canadian legislation does provide the Government of Canada with ample authority to give essentially the same rights to the United States as are provided by Section 115.*" [Emphasis added]. Costle letter to Muskie

47/ Costle letters to Muskie.

48/ *Id.*

49 Memorandum from David Hawkins, EPA Assistant Administrator for Air, Noise and Radiation to Walter Barber, EPA Office of Air Quality Planning and Standards, January 13, 1981.

50/ In response to a request under the Freedom of Information Act, EPA explained that there were no records regarding any staff responses to Hawkins' memo to the Office of Air Quality Planning and Standards. Letter from Dennis Tirpak, EPA Director, Office of Exploratory Research to Robert Blacher, attorney for the Province of Ontario, August 4, 1981.

51 In *Ohio v EPA* Nos 81-1310, 1311, 1312 (D.C. Cir, 1981) the State of Ohio and two Ohio utilities filed petitions pursuant to Section 307(b)(1) of the Clean Air Act in the D.C. Circuit to review and set aside "the purported final action of the Administrator" disclosed in the EPA press release of January 16, 1981. The Province of Ontario moved to intervene "in order to support the action of the Administrator in making the threshold findings which initiate proceedings leading to reduction in United States pollutant emissions threatening the Canadian environment." Province of Ontario Response in Opposition, pp. 3-4.

However, the Court never reached the issue before it; namely whether the press release was a final action within the purview of Section 307(b)(1). Early in the litigation, all parties essentially agreed to dismissal. Ohio agreed to voluntarily withdraw its petition, and all other parties agreed, in essence, that the case should be dismissed as not being ripe for review. In addition, EPA and Ontario specifically stipulated that:

> "dismissal of the cases shall not be construed by the parties to this stipulation as having any legal significance or affect on any future litigation between the parties to this stipulation regarding Section 115 of the Clean Air Act, 42 USC § 7415." Stipulation between EPA and Province of Ontario, *Ohio v. EPA*, D.C. Cir. No. 81-1310, *et al.*, October 19, 1981.

With motions for dismissal and the stipulation before it, the Court dismissed the petitions without opinion. Thus the Court never reached the merits of the legal significance of the press release itself nor addressed the issue of whether the Administrator had made findings as required by Section 115(a) and (c) to trigger a duty to notify the states pursuant to Section 115(a).

52/ Congress in Section 304 eliminated the customary barriers to standing -- that the plaintiff had experienced an "injury in fact" -- in order to encourage "private attorneys-general" to ensure enforcement of the Act. See e.g., *Metropolitan Coalition for Clean Air v District of Columbia*, 511 F2d809, 814 and note 26 (C.A.D.C. 1975).

53/ It can be argued that foreign governments are considered to be "persons" under Sections 302(e) and 304. Section 302(e) could be construed as inclusive of possible litigants rather than defining the complete set. Such a construction was placed on a similarly-worded definition under the antitrust laws and thus the definition in the statute was not a mandate to reverse the longstanding presumption that foreign nations are entitled to sue in courts of the U.S. *Pfizer, Inc. v Government of India* 434 U.S. 308 (1978) Accordingly the Court in *Pfizer* found that India was a "person" within the meaning of Section 4 of the Clayton Act in spite of the fact that the section did not include foreign governments in the list of those authorized to sue.

54/ Secretarial Decision Paper, Reciprocity Status of Canada, February 2, 1982; See also, Memorandum to the Secretary of Interior James Watt, from Associate Solicitor, Energy and Resources, Re: Determinations of Reciprocity under Section 1 of the Mineral Lands Leasing Act of 1920 (30 U.S.C. § 181), February 2, 1982.

55/ Air pollution is defined as:

> (a) condition of the ambient air, arising wholly or partly from the presence therein of one of more air contaminants, that endanger the *health, safety, or welfare of persons*, that interferes with the normal enjoyment of life or property, that endangers the health of animal life, *or that causes damage to plant life* or property. (Emphasis added)(Canada Clean Air Act §21.1(1)(b).

56/ Since the objective of the Canadian Parliament in enacting Section 21.1 was to assure reciprocity with the U.S. Section 115, it is clear that the phrase was *intended* to encompass environmental effects among others. House of Commons Debates, Vol. 124, No. 115, 1st Session, 32nd Parliament, December 10, 1980.

57/ Statement of Raymond Ewing, Deputy Assistant Secretary of State for European Affairs, Hearing before the House Subcommittee on Human Rights and International Organizations and on Inter-American Affairs of the Committee on Foreign Affairs, May 20, 1981.

58/ Fairley, H. "Private Remedies for Transboundary Injury in Canada and the United States: Constraints upon having to see where you can collect." 10 *Ottawa L. Rev.* 253, 254 (1978); Arbitblit, D. "The Plight of American citizens injured by transboundary river pollution," 8 *Ecology L.Q.* 339 (1979); McCaffrey, S., "Transboundary pollution injuries: jurisdictional considerations between Canada and the United States," 3(2) *Calif. West Int'l L.J.* 191 (1973), Cudahey, R., "Clouds on the Horizon: Acid rain in domestic courts," address at conference on the environmental consequences of energy production, Banff, Alberta (March 21, 1981); and Swaigen, J., "Compensation of pollution victims in Canada," Draft Report to the Wastewater Research Center for the Regulation Reference, Economic Council of Canada (January 1980).

59/ U.S. Circuit Judge Richard Cudahy wryly commented at a recent conference on transboundary pollution that "domestic judges are not well versed in diplomacy, and we often seem to handle sensitive international issues with the delicacy of a fleet of bulldozers rumbling across the perma-frost." Remarks of Judge Richard Cudahy, presented at the Conference on the Environmental Consequences of Energy Production, Banff, Alberta (March 12, 1981).

[60] For detailed discussion of nuisance actions as a vehicle for recovery of pollution related damages see Rice, "Pollution as a Nuisance: Problems, Prospects, and Proposals," 34 Amer. Trial Lawyers J. 203; Morrissey, "Private Nuisance: A Remedy Against Air Pollution," 34 Insur. Counsel J. 367 (1971); Note, Private Nuisance Law: Protection of the Individual's Environmental Rights," 8 Suffolk Univ. Law Review 1162 (1974).

[61] Michie v. Great Lakes Steel Division, 495 F.2d 213 (6th Cir. April 4, 1974).

[62] Ohio v. Wyandotte Chemical Corp. No. 904, 571 (Ct. App. Ohio April 24, 1975).

[63] Wetstone, note 1, *supra* at 50003 n. 26. See also, Comment, "Seventh Circuit Interprets Federal Common Law of Nuisance to Authorize Municipalities to Sue for Damages" 9 ELR 10168 (1979).

[64] Milwaukee v Illinois, 49 U.S.L.W. 4445, 11 ELR 20406 (April 28, 1981).

[65] Middlesex County Sewerage Authority v. National Sea Clammers Association, 49 U.S.L.W. 4783, 11 ELR 20684 (June 25, 1981).
For a more detailed discussion of these cases and their impacts on the federal common law see Comment, "Requiem for the Federal Common Law of Nuisance," 11 ELR 10191 (1981).

[66] However, in *New England Legal Foundation v. Costle*, 16 ERC 1829 (2d Cir. 1980), the Second Circuit, while refusing to reach the "broad question of whether the Clean Air Act totally preempts federal common law nuisance actions based on the emissions of chemical pollutants into the air", states, in dicta:

> We note that there are similarities between the Water Pollution Control Act, which was the subject of *City of Milwaukee*, and the Clean Air Act, which is here involved. Arguably, such similarities might justify a broad prohibition against federal common law nuisance actions such as the instant one. See e.g., *Ethyl Corp. v. EPA*, 541 F.2d 1 17 (D.C. Cir.), *cert. denied sub nom. National Petroleum Refiners Assn. v. EPA*, 426 U.S. 941 (1976). We also note, however, that the Clean Air Act differs substantially from the Water Pollution Control Act in areas which the majority of the Court in *City of Milwaukee* found were especially significant but which bear no relation to the facts herein. For example, Justice Rehnquist, writing for the majority in *City of Milwaukee*, found it especially significant that under the Water Pollution Control Act the EPA regulated *every* point source of water pollution. 451 U.S. at ___, 49 U.S.L.W. at 4449. (Emphasis in original.) Under the Clean Air Act, in contrast, the states and the EPA are not required to control effluents from every source, but only from those sources

which are found by the states and the agency to threaten national ambient air quality standards. 42 U.S.C. §7410(a)(2)(d). In light of those and other differences, we leave for a more appropriate case the question of whether all federal common law nuisance actions involving the emission of chemical pollutants into the air are precluded by the statutory scheme set forth in the Clean Air Act. *Id.* at 1853, note 2.

67/ Several commentators have undertaken detailed analyses of the procedural nuances of transboundary tort actions concerning pollution related damages. See Fairley, Arbitbilt, and McCaffrey, note 102, *supra*.

A plaintiff in a transboundary tort action may bring suit in the foreign jurisdiction where the source is located, or sue in his own nation at the site of the damage. In the latter situation, the plaintiff may find that the foreign court will not take jurisdiction over cases alleging harm to foreign property. If the former route is chosen, the plaintiff may discover that the court is unable to enforce the judgment against a foreign defendant, unless that defendant has assets in the court's jurisdiction.

Where the plaintiff seeks redress for an injury to real property, it is easier to bring a transboundary tort action in the U.S. courts than in Canada. Canadian courts subscribe to the "local action rule" of *British South Africa v. Companhia de Mogambique*, (1893) A.C. 602, 63 L.J.Q.B. 70 H.L..), rev'g (1892) Q.B. 358 (C.A.), which denies subject-matter jurisdiction over actions involving damage to real property located in another state. U.S. courts, however, follow the precedent in *Bulwer's Case*, 7 Coke 1a, 177 Eng. Reprints 411 (C.P. 1584), which allows jurisdiction if the acts causing the injury to property occurred in the forum state. However, if the action is based upon a personal right, courts in both nations will assume jurisdiction.

68/ Helsinki Rules on the Uses of the Waters of International Rivers, International Law Association (London, 1967) reprinted in Barros & Johnston, note 128 *supra*.

69/ See *e.g.* Levin, *Protecting the Human Environment, Procedures and Principles for Preventing and Resolving International Controversies* (United Nations Institute for Training and Research, 1977).

70/ The most respected of international adjudicatory bodies, the International Court of Justice, may rule on a case only after the involved nations have consented to a reference, a rare occurrence. See Statute of the International Court of Justice, arts. 36, 37.

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